TITTABAWASSEE RIVER FLOODPLAIN SCREENING-LEVEL ECOLOGICAL RISK ASSESSMENT

POLYCHLORINATED DIBENZO-*P*-DIOXINS
POLYCHLORINATED DIBENZOFURANS

SUBMITTED TO: MICHIGAN DEPARTMENT OF ENVIRONMENTAL QUALITY,
REMEDIATION & REDEVELOPMENT DIVISION,
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SUBMITTED BY: GALBRAITH ENVIRONMENTAL SCIENCES LLC.,
NEWFANE, VERMONT

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LIST OF ACRONYMS

DW	Dry Weight
ERA	Ecological Risk Assessment
GES	Galbraith Environmental Sciences LLC
HI	Hazard Index
HxCDF	Hexachlorodibenzofuran
Kcal	kilocalorie
LD50	Lethal Dose that killed 50% of test organisms
LOAEL	Lowest Observed Adverse Effect Level
MDEQ	Michigan Department of Environmental Quality
NOAEL	No Observed Adverse Effect Level
NWR	National Wildlife Refuge
PCB	Polychlorinated Biphenyl
PCDD	Polychlorinated Dibenzo-*p*-dioxin
PCDF	Polychlorinated Dibenzofuran
PeCDF	Pentachlorodibenzofuran
PCH	Polychlorinated Hydrocarbon
Pg/g	Picograms/gram
PCOC	Potential Contaminant of Concern
TEF	Toxicity Equivalence Factor
TCDD	Tetrachlorodibenzo-*p*-dioxin
TCDF	Tetrachlorodibenzofuran
TCDD-EQ	TCDD-Equivalent
TRV	Toxicity Reference Value
UF	Uptake Factor
U.S. EPA	United States Environmental Protection Agency
USFWS	United States Fish and Wildlife Service
WHO	World Health Organization
WW	Wet weight

EXECUTIVE SUMMARY

Risks to six species of birds and mammals from consuming soils and invertebrate, mammalian and avian prey from the floodplain of the Tittabawassee River downriver of Midland were evaluated using a screening-level ecological risk assessment. This analysis was based on empirical soil PCDD/PCDF concentrations and bioaccumulation, toxicological, and ecological data from the scientific literature. The question addressed by this ecological risk assessment was: can unacceptable risk to ecological receptors in the Tittabawassee River floodplain be reasonably discounted?

The results of this analysis show that:

- Using empirical soil PCDD/PCDF data and assuming soil-organism uptake factors from the scientific literature, TCDD-EQ concentrations in invertebrates, small mammals and birds in the Tittabawassee River floodplain downriver of Midland are predicted to average 393, 12,048, and 6,038 pg/g TCDD-EQ, respectively (using WHO avian TEFs), and 124, 5,083, and 2,552 pg/g (using WHO mammalian TEFs). These organisms are assumed to be the prey of the six receptor species.

- The majority of the TCDD-EQ in the invertebrates, small mammals and birds is predicted to be contributed by two congeners, 2,3,7,8-TCDF and 2,3,4,7,8-PCDF.

- Food chain models predict that the daily TCDD-EQ intake rates for the six receptor species are:

Red fox	1,732,613 pg
Short-tailed shrew	1,049 pg
Red-tailed hawk	1,586,547 pg
American kestrel	318,013 pg
American woodcock	90,586 pg
American robin	46,879 pg

- Protective toxicity reference values (daily TCDD-EQ doses) for the six receptor species were established as:

Red fox	2,050 pg
Short-tailed shrew	38 pg
Red-tailed hawk	17,136 pg
American kestrel	1,820 pg
American woodcock	3,080 pg
American robin	1,134 pg

- Combining the TCDD-EQ intake rates with the toxicity reference values resulted in the following hazard indices:

Red fox	845
Short-tailed shrew	28
Red-tailed hawk	93
American kestrel	174
American woodcock	29
American robin	41

  All of these hazard indices (based on soil mean PCDD/PCDF concentrations) represent unacceptable risk to the receptor species

- Hazard indices were also calculated based on soil median, maximum and upper 95% confidence limits of the mean. All of these also showed unacceptable risk to each of the receptors and ranged up to 6,636 for the most at-risk species (red fox) and 220 for the least at-risk species (short-tailed shrew).

The main conclusion of this screening-level ERA is that the possibility of unacceptable risks to terrestrial receptors in the Tittabawassee River floodplain due to soil contamination by PCDDs and PCDFs cannot reasonably be discounted. Indeed, the relatively high HI values obtained may be an indication that it may be more likely than not that risk actually pertains in the assessment area. However, further site-specific studies are needed before any such risks can be confirmed or rejected.

1. INTRODUCTION

In January 2003, Michigan Department of Environmental Quality (MDEQ) commissioned Dr. Hector Galbraith of Galbraith Environmental Sciences LLC (GES) to carry out an evaluation of risks to ecological resources posed by polychlorinated hydrocarbon (PCH) contaminants in the Tittabawassee River and its floodplain. The risks posed by PCHs in the aquatic environment have already been reported (GES, 2003). This document preliminarily evaluates the magnitude of risks posed to terrestrial ecological receptors exposed to PCHs in the Tittabawassee River floodplain.

Fewer data exist for the terrestrial environment of the Tittabawassee River floodplain than for the aquatic environment of the Tittabawassee River. Because of this, the uncertainty associated with risk predictions is necessarily greater and this ERA should be regarded as a screening-level assessment, addressing the main question: can we reasonably ignore the possibility of unacceptable risk to wildlife in the assessment area? The distinctions between screening-level and more definitive ERAs are explored in greater detail in Section 2.1 of this report.

1.1 Report Structure

Chapter 2 of this report comprises a general introduction to the objectives and process of ecological risk assessment (ERA). It discusses in detail the ERA process framework developed by U.S. EPA (U.S. EPA, 1998), since that is the methodological approach followed in this evaluation, and the role of screening-level assessments. Chapters 3 through 5 detail the results of the various components of the ERA (*Problem Formulation*; *Analysis*; and *Risk Characterization*). Chapter 6 discusses uncertainty associated with this ERA. Chapter 7 considers the implications of the results of this and the previous ERA (GES, 2003) for organisms that could use both the aquatic and terrestrial habitats in the assessment area. Chapter 8 lists the scientific references used in the development of this ERA.

1.2 Units

Throughout this report use is made of concentrations of contaminants in biotic and abiotic media. These are typically expressed as picograms/gram (pg/g). 1 pg/g is also equivalent to a part per trillion (ppt). When the concentrations refer to soils the units are in dry weight (dw); when they refer to biota concentrations they are in wet weight (ww), unless otherwise stated.

2. ERA – OBJECTIVES AND PROCESS

Typically, the objectives of ERA include being able to predict the likelihood that environmental stressors may pose risks to ecological resources, to anticipate where and when such risks are most likely to occur, and to determine the types and magnitudes of effects. The information obtained through ERA can then be used to help inform and focus mitigation strategies, or to help quantify trade-offs and ecological costs and benefits among alternative response actions. As an analytical problem-solving approach, ERA has mostly focused on the risks that may be posed to ecological resources by chemical contaminants, although, it can also be used to evaluate the potential risks posed by non-chemical stressors.

In essence, ERA compares measured or predicted degrees of stress on organisms or ecological systems with benchmark values that are believed, or known, to result in one or more levels of effect on the exposed organism or system. When the stressors are chemical contaminants, the process becomes one of comparing the level of stressor to which the organism(s) is exposed (the exposure concentration) to a protective toxicological benchmark established through either laboratory studies or in the field (Bartell *et al*., 1992; Calabrese and Baldwin, 1993; NRC, 2001; U.S. EPA, 1998). The ratio derived from this comparison is an index of the probability and magnitude of risk to the exposed organism(s).

While the overall approach of ecological risk assessment may be as simple as outlined above, in any actual ERA a number of assumptions may have to be made about (for example) the level of exposure, the sensitivity of the target organisms to the contaminants, the fate and transport of the contaminants, the effects of multiple contaminants, or the actual responses of the organisms to exposure. Often, contaminant-, organism-, or site-specific data do not exist and values may have to be assumed from other contaminants, organisms, or sites. Because of such assumptions, uncertainty will be associated with the parameters and results of an ERA. In recognition of this, and to facilitate and encourage consistency in the way that ERAs are performed, the U.S. EPA has developed a process framework and guidelines (U.S. EPA, 1998). The 1998 EPA guidelines were developed by EPA staff and were extensively reviewed and modified by expert practitioners. The resulting framework and set of guidelines are now widely regarded as the "industry standards" for conducting ERA in the United States. The U.S. EPA (1998) framework comprises the risk assessment approach used in this ERA and is described in Section 2.2 of this report.

2.1 Screening-level ERA

The ERA reported in GES (2003) utilized site specific data including sediment and biota (fish and wildlife) contaminant concentrations. Because of the uncertainty removed by the existence of these biological data, GES (2003) was able to address the questions: does risk exist in the aquatic environment, and what is its magnitude and distribution? Fewer data are available for the terrestrial environment of the Tittabawassee River floodplain and the uncertainty associated with risk predictions is necessarily greater. Consequently, this ERA should be regarded as a screening-level assessment, and does not address questions about actual risk magnitudes or spatial distribution. Rather, it addresses the more preliminary question: can we reasonably disregard the possibility of unacceptable risk to wildlife in the assessment area due to contaminants in the floodplain? If the answer to this question is no, it need not necessarily mean that risk exists. It does mean, however, that the possibility of risk cannot be ignored and that further studies are necessary to reduce uncertainty to a degree where more definitive statements about the existence or lack of risk can be made. If the answer is yes, given the degree of protectiveness built into the process it may be assumed that the possibility of unacceptable risk can be reasonably ignored.

In screening-level ERAs, it is important to avoid the likelihood of making Type II or false negative errors (i.e., conclusions of no risk when, in fact, risk does exist). To avoid this, the assumptions that are typically used in such ERAs are typically more protective of the exposed wildlife than those that would be included in ERAs with less uncertainty.

2.2 The U.S. EPA (1998) Ecological Risk Assessment Framework

Figure 2-1 shows a simplified form of the U.S. EPA (1998) ERA framework. It comprises three main components or stages:

Problem Formulation. In this stage the potential risk issues at the site(s) are identified, the objectives of the ERA are articulated, and an analysis plan developed. To effectively identify and describe the potential risk issues, existing information on the potential contaminants of concern (PCOCs), the exposed receptors, the types of toxicological responses that may occur due to exposure to the PCOCs, and the environmental factors that may modify the fate and transport and toxicology of the PCOCs or the behavior of the receptors are gathered and combined into a conceptual model of the site. The function of the conceptual model is to preliminarily identify and link the important components of the system, its processes, potentially exposed receptors, and the PCOCs that may result in risk to those receptors. Also identified through this process are assessment endpoints for the ERA. Assessment endpoints are expressions of the ecological resources that are to be protected and that are the focus of the ERA.

Overall, therefore, the Problem Formulation stage defines the scope, terms, and direction of the subsequent ERA. It is important to note that the whole ERA process is often iterative and the products of the Problem Formulation stage (i.e., conceptual models, assessment endpoints, and analysis plans) may be altered as data are collected during the next stage.

Analysis. The Analysis stage of an ERA has two main objectives:

1) To characterize and quantify the exposure of the ecological receptors to the PCOCs. Exposure may be quantified at several different levels according to the environmental behavior and toxicology of the contaminants. Contaminant concentrations in the diet, dietary intake rates, and receptor tissue concentrations are all often used as measures of exposure. In screening-level ERA actual tissue residue data may be sparse or lacking, and exposures to receptors may have to be modeled from environmental matrices (soils or sediments, for example) using parameters from the scientific literature.

2) To characterize and quantify the types and degrees of toxicological responses that may occur among the receptors on exposure to the contaminants, and the sensitivities of these receptors. The outcome of this component of the Analysis stage is a stressor-response profile that addresses the following questions: what exposures to the contaminant are likely to induce toxicological responses, and what exposures are protective of the organism?

The exposure and stressor-response information is combined in the final stage of the ERA (Risk Characterization) into an assessment of the level of risk that may be (or is being) incurred.

Risk Characterization. In this final stage of the ERA the products of the Analysis stage are combined to derive an estimate of risk to the exposed receptors. In screening-level ERA such estimates are typically point estimates, as when the ratio between the exposure and some response level is calculated. This estimate is termed a Hazard Index (HI). An HI greater than 1 indicates that the exposure level exceeds the response threshold and, therefore, the risk of that response has been incurred. The more that the HI exceeds 1 the greater the risk of more severe effects: an HI that equals or only just exceeds 1 might signify that a risk exists to individual organisms, while HIs larger than that might indicate risks to larger components of the population or community.

2.3 Uncertainty

Uncertainty is inherent in all ecological risk assessments, but especially so in screening-level assessments. It may arise from a large number of sources but is often due to site-specific information on food chain contamination not being available, or that the stressor response information is not complete and assumptions must be made, or that no stressor-response information exists for the receptor and a surrogate species must be used. Regardless of its source or type, the ERA must explicitly recognize and accommodate this uncertainty. Uncertainty factors may be applied in HI-based ERAs in (for example) inter-species conversions or levels of effects conversions. One of the main objectives of any ERA should be to acknowledge its inherent uncertainty and then reduce that uncertainty to the extent possible. If it is not possible to reduce uncertainty to a level considered acceptable by the risk assessors and managers, the ERA must reflect this in its statements regarding the magnitude or spatial distribution of risk.

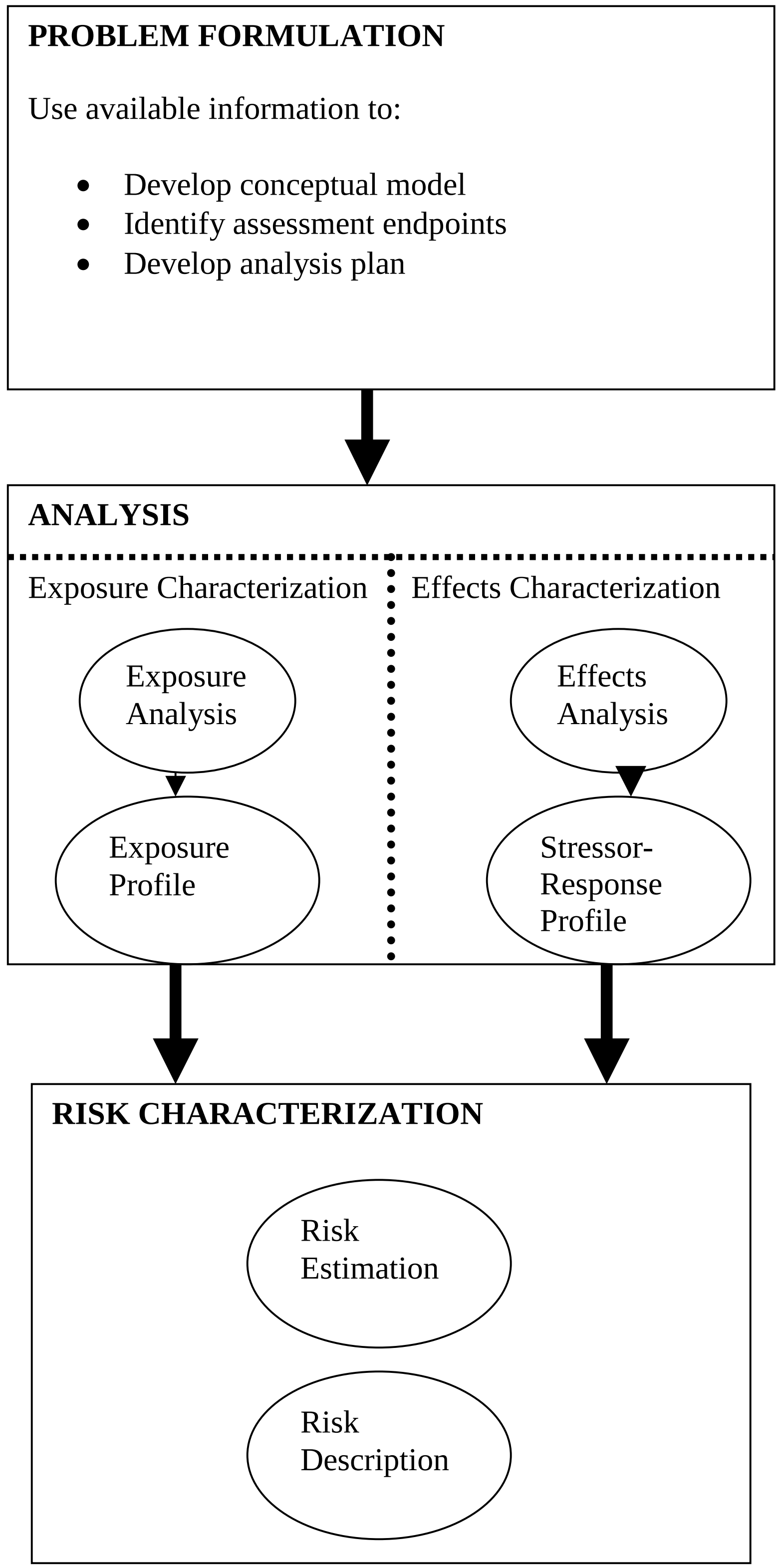


Figure 2-1. Simplified schematic of the U.S. EPA (1998) ERA framework.

3. PROBLEM FORMULATION

3.1 The Assessment Area

Beginning in Roscommon and Ogemaw Counties in north-central Michigan, the Tittabawassee River flows south and southeast for a distance of approximately 80 miles to its confluence with the Saginaw River, which then flows east into Saginaw Bay on Lake Huron (Figure 3-1). For the first 60 miles of its course, the Tittabawassee River flows through a largely rural and agricultural landscape. Major tributaries in this upriver reach are the Tobacco, Pine, and Chippewa Rivers. At the city of Midland, the predominant land-use changes as the river flows past major chemical manufacturing and processing plants, industrial and municipal wastewater discharges, and areas of urban land-use. Downriver of Midland, the river and its floodplain once again become largely rural in character, with land-use split among residential, agricultural, public parks, and other protected areas, including the Shiawassee National Wildlife Refuge (NWR). At its confluence with the Saginaw River near the City of Saginaw, the land-use near the river once again becomes largely urban and industrial.

The distance from Midland to the confluence at Saginaw is approximately 22 miles and throughout most of that length, the river flows through a well-marked floodplain. In most years, parts of the floodplain are flooded by the river. In wet years, the majority of the floodplain may be inundated. Thus, the river is hydrologically connected to the floodplain on a regular basis. This floodplain (defined as the 100-year floodplain) between Midland and the confluence at Saginaw is approximately 9.6 square miles in extent (N. Ekel and A. Brouillet, MDEQ, *pers comm*).

Downriver of the City of Midland, the floodplain provides habitats suitable for a large variety of wildlife species. These habitats include deciduous forests and shelterbelts, wetlands, and agricultural areas. Because of this habitat diversity, the diversity of wildlife is high. For example, over 220 species of birds have been recorded at the Shiawassee NWR alone (http://www.npsc.nbs.gov/resource/othrdata/chekbird/r3/SHIAWA.HTM). This list includes carnivorous birds such as red-tailed hawks (*Buteo jamaicensis*) and screech (*Otus asio*) and great horned owls (*Bubo virginianus*), and insectivorous birds such as American robins (*Turdus migratorius* -seen on field trips to the assessment area) and American woodcocks (*Scolopax minor*). It should also be noted that the avian species list for the Shiawassee NWR probably under-represents the actual avian biodiversity for the entire floodplain since habitats that are well represented on the rest of the floodplain (e.g., agriculture and urban) are not found on the NWR.

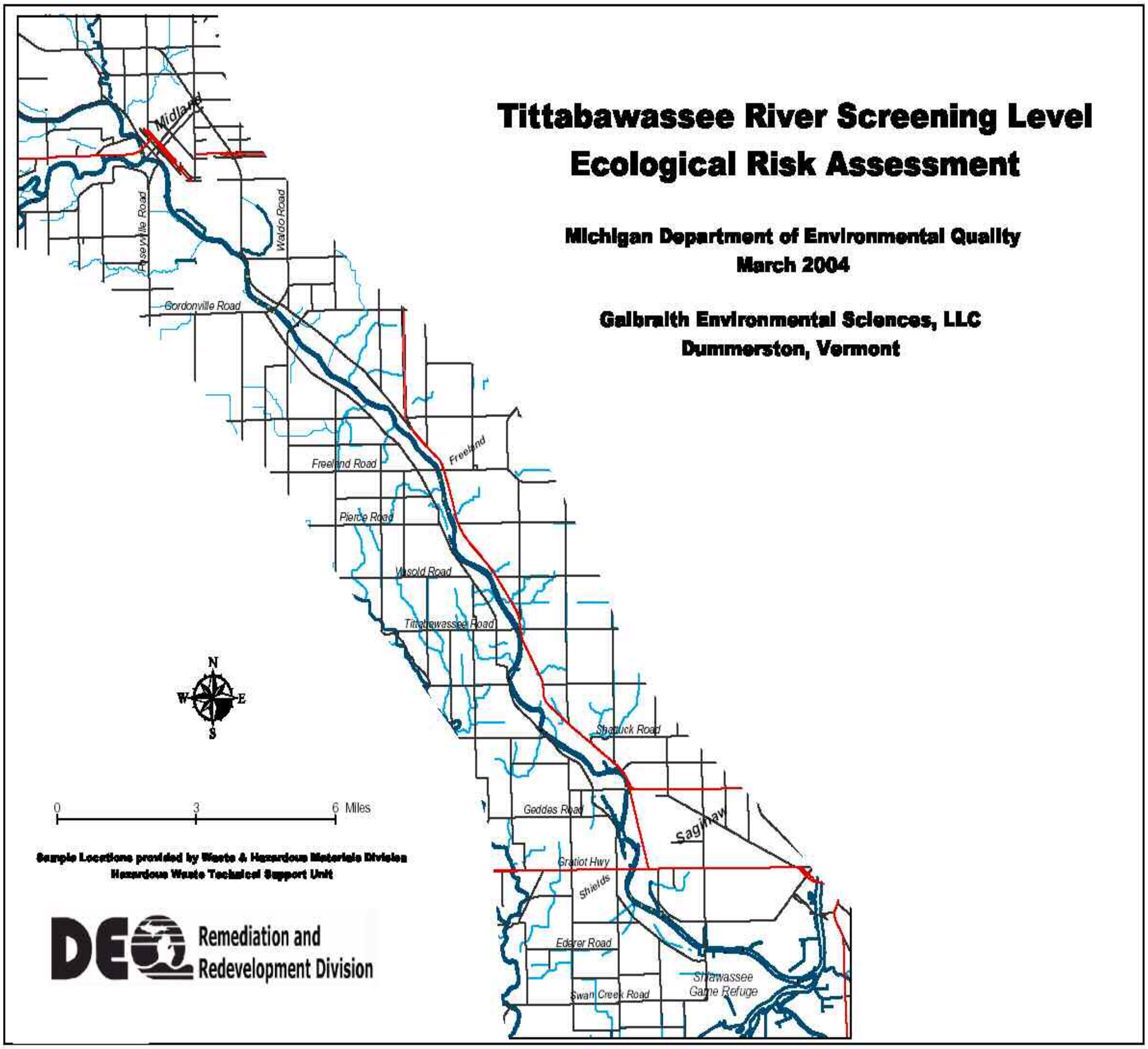


Figure 3-1. Map of the assessment area from Midland to the confluence of the Tittabawassee and Saginaw Rivers.

Chemical manufacturing operations began along the Tittabawassee River in the City of Midland during the 1890's. It is possible that chemical manufacturing operations with the potential to generate PCH-contaminated waste material could have begun as early as the 1920's or 1930's. However, even after the enactment of the Michigan Water Resources Commission Act in 1929, and the federal Clean Water Act in 1972, very little was known about the amount and type of PCH compounds that were being released to the air, soil, or the Tittabawassee River from these industrial operations.

The implementation of state and federal environmental regulatory programs has historically focused on controlling, reducing, and eliminating PCH releases at the source. However, during 1984 the U.S. EPA collected floodplain soil samples at and near the Shiawassee National Wildlife Refuge, located approximately twenty miles downstream of Midland. These samples identified elevated concentrations of polychlorinated dibenzo-*p*-dioxins (PCDDs) and polychlorinated dibenzofurans (PCDFs). Although an initial indication of a more widespread contamination problem, state and federal regulatory efforts continued to focus on reducing, eliminating, or controlling sources of PCDD/PCDF releases.

During April of 2000, floodplain soil samples were collected during the development of a wetland construction project located at the confluence of the Tittabawassee and Saginaw Rivers. Elevated PCDD/PCDF concentrations were measured. Subsequent confirmation samples collected by the MDEQ discovered TCDD-EQ concentrations as high as 7,300 pg/g, over 80 times the residential direct contact criterion (RDCC) of 90 pg/g established under Part 201, Environmental Remediation, of the Michigan Natural Resources and Environmental Protection Act, 1994 PA 451, as amended, and the Part 201 administrative rules. Concern over the public and environmental health implications of these sample results prompted the MDEQ to develop and implement a phased floodplain soil sampling and assessment program in the Tittabawassee River floodplain to determine the source and extent of the contamination.

The Phase I portion of the soil sampling program was implemented during December 2000 through July 2001. The MDEQ collected 34 soil samples from five locations within a two-mile stretch of the Tittabawassee River floodplain located near the City of Saginaw. The Phase I sampling effort confirmed the presence of elevated PCDD and PCDF concentrations within the lower Tittabawassee River floodplain near the river's confluence with the Saginaw River. Only seven of the 34 samples contained TCDD-EQ concentrations less than the Part 201 RDCC.

The MDEQ also collected and analyzed sediment samples from the Tittabawassee, Chippewa, and Pine rivers, during spring/summer 2001. The objective of the MDEQ Sediment Study was to characterize concentrations of contaminants in Tittabawassee and tributary river sediments both upstream and downstream of Midland. PCDDs and PCDFs were analyzed as part of this study. Surface sediment samples were collected from the tributaries and the Tittabawassee River beginning immediately upstream of Midland and continuing downstream to its confluence with the Saginaw River. Sediment cores were also collected in selected areas, together with some floodplain soil samples. These samples confirmed that elevated concentrations of PCDDs and PCDFs are pervasively present in sediment and floodplain soil downstream of Midland.

An expanded Phase II floodplain soil sampling program was implemented by the MDEQ during 2002. Floodplain soil samples were collected from sixteen locations extending from eight miles upstream of Midland downstream to shoreline areas located along the Saginaw River and the inner portions of Saginaw Bay. Phase II results confirmed that PCDD/PCDF contamination of flood plain soil is extensive, extending downstream from Midland. The highest concentrations were consistently observed within the twenty-two miles of the Tittabawassee River floodplain downstream of Midland. Elevated PCDD/PCDF concentrations were identified at one location to a depth of four feet below the ground surface.

The MDEQ Phase I and II floodplain sampling initiatives focused on public parks and other public access areas. During the fall of 2003 the MDEQ obtained permission from 22 residential property owners to collect and analyze soil samples for analysis. The residential properties were located near and along the river shoreline downstream of Midland. The sample results identified extensive PCDD/PCDF contamination of flood-prone portions of the residential properties downstream from Midland, and confirmed the findings of the Phase I and Phase II sampling programs.

During the summer of 2003 the State of Michigan entered into a waste management license with the Dow Chemical Company (Dow) under the authority of the federal Resource Conservation and Recovery Act (RCRA). The license provided for the investigation, interim response, and full remediation of dioxin contamination, and other contamination, that may have been released from the Dow Midland manufacturing complex to the Tittabawassee River sediment and floodplain soil, as well as Midland soil. Dow is in the initial stages of implementing license conditions, but, as of the date of this report, this implementation effort has not provided any new information that can be used for the purpose of evaluating ecological risk presented by PCDD/PCDF contamination of flood plain soil.

Analyses performed by GES (GES, 2003) showed that the levels of PCDD/PCDF contamination in the aquatic environment of the Tittabawassee River pose risks to piscivorous birds and mammals that ingest fish prey from the river. This companion ERA focuses on the terrestrial environment of the Tittabawassee River floodplain between Midland and the confluence with the Saginaw River.

3.2 Contaminants in the Assessment Area

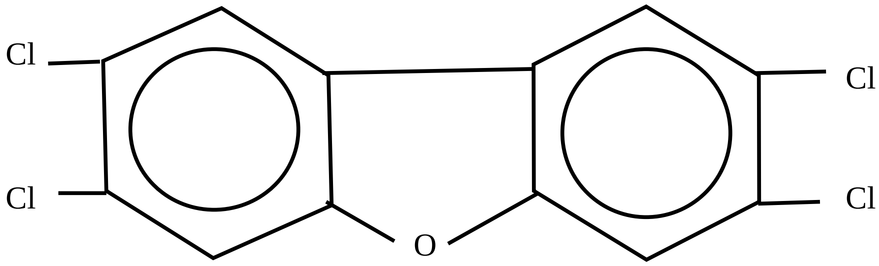
Floodplain soil and riverbed sediment sampling and analysis have shown that beginning at Midland and extending downriver to the confluence with the Saginaw River the Tittabawassee River and its floodplain are contaminated to above background levels with PCHs. Upriver of Midland, PCH concentrations in the floodplain soil are either low or non-detectable (MDEQ, 2002; MDEQ, 2003). The soil and sediment samples collected by the MDEQ from the Tittabawassee River and its floodplain upstream and downstream of Midland indicate that polychlorinated biphenyls (PCBs) are not present in high concentrations. Similarly, pesticides were detected at only a few sample sites and at very low concentrations (MDEQ, 2002).

Unlike PCBs and pesticides, PCDDs and PCDFs were found at elevated concentrations in all floodplain soil or sediment samples that were collected downriver of Midland (MDEQ, 2002; MDEQ, 2003). Much lower or non-detectable concentrations were found upriver of Midland. In an earlier study, Amendola and Barna (1986) also reported PCDD concentrations at up to 16,000 pg/g in the Tittabawassee River sediments downriver of Midland, but did not detect PCDDs upriver of Midland. That these contaminants were also being accumulated in foodchains was confirmed by analyses of fish collected by MDEQ from the Tittabawassee River in 2002, from analysis of the eggs of chickens (*Gallus domesticus*) foraging in the floodplain in 2002 (MDEQ, 2003), and from analyses of the eggs of wood ducks (*Aix sponsa*) and hooded mergansers (*Lophodytes cucullatus*) nesting in the Shiawassee National Wildlife Refuge in 2003 (USFWS and MDEQ unpublished data reported in GES, 2003). The wood duck and hooded merganser samples contained higher concentrations of PCDDs and PCDFs than in reference areas elsewhere in Michigan.

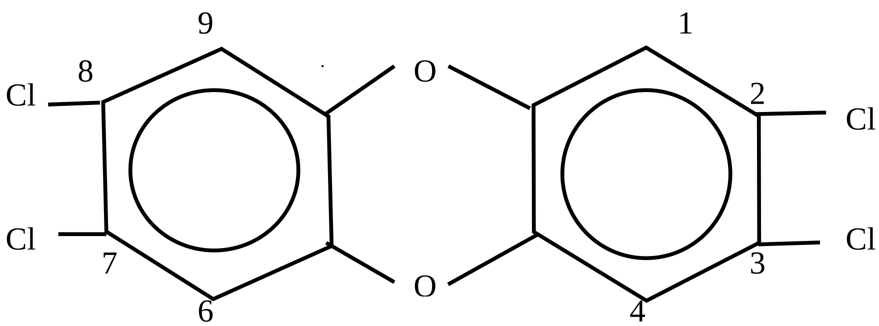
GES (2003) showed that the PCDDs and PCDFs in the Tittabawassee River sediments pose risks to piscivorous predators. The purpose of this ERA is to determine the likelihood that risks may also be posed to organisms in terrestrial food chains from the PCDD/PCDFs in the floodplain soils.

3.2.1 Structure, toxicity, and environmental behavior of PCDDs and PCDFs

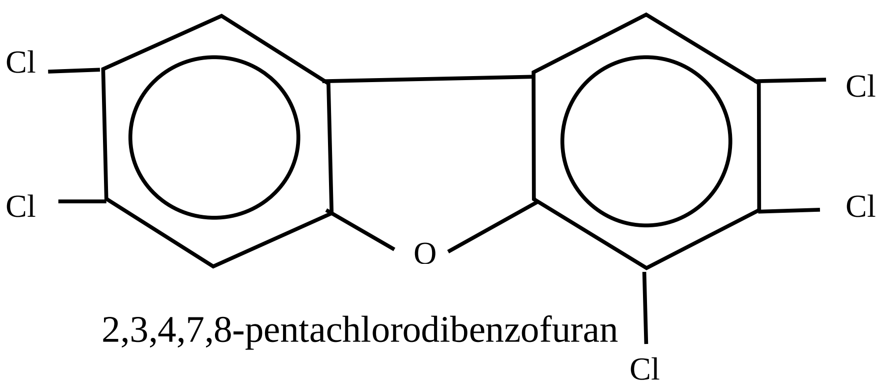
PCDDs and PCDFs are classes of compounds consisting of large numbers of individual isomers or congeners (75 and 135, respectively). The skeleton of the PCDD molecule consists of two phenyl rings joined by two oxygen bridges. That of the PCDF molecule comprises two phenyl rings joined by one oxygen bridge and one single bond (Figure 3-2). The individual congeners of PCDDs and PCDFs differ in their patterns of chlorine substitution; examples are shown in Figure 3-2. The degree and pattern of substitution affects the stereochemistry of the congener, and is responsible for inter-congener differences in environmental behavior and toxicity. Congeners that are substituted only at the 2, 3, 7, or 8 positions (Figure 3-2) are lipophilic, structurally rigid, and resistant to environmental degradation. They also readily bind to the crucial AhR enzyme receptor in vertebrates, the molecular event that is responsible for the adverse toxicological effects of many PCDDs and PCDFs (Bosveld, 1995; NRC, 2001; Safe, 1993; Van den Berg *et al*., 1994). Because they are lipophilic and resist degradation and metabolism, they readily



2,3,7,8-tetrachlorodibenzofuran



2,3,7,8-tetrachlorodibenzo-*p*-dioxin



2,3,4,7,8-pentachlorodibenzofuran

Figure 3-2. Molecular structures of PCDD and PCDF congeners

bioaccumulate in food chains and may biomagnify at successive trophic levels. 2,3,7,8-TCDD is one of the most toxic compounds yet tested, eliciting adverse effects in some organisms at dose concentrations in the parts per trillion range (Murray *et al*., 1979; Nosek *et al*., 1992; U.S. EPA, 1993a). Congeners without at least the 2,3,7, and 8 positions substituted are less stable, more susceptible to degradation by metabolic and environmental processes, and bind less readily to the AhR receptor. In this report PCDDs and PCDFs will mean 2,3,7,8-substituted isomers, unless otherwise noted.

Because of their toxicologies, biochemistries, and environmental chemistries, PCDDs and PCDFs can pose risks to ecological receptors at relatively low exposures. Organisms at the tops of food chains (i.e., vertebrate predators) generally experience higher levels of exposure than those at lower trophic levels. Also, early life stages of organisms are more sensitive than older life stages. Thus, their adverse effects in laboratory and free-ranging populations are most often manifested in the young or embryos of top predators (e.g., Van den Berg *et al*., 1994; Giesy *et al*., 1994; Nosek *et. al*., 1993; Powell *et al*., 1996; White and Hoffman, 1995. Summaries in: Eisler, 1986; Hoffman *et al*., 1996).

Different PCDD and PCDF congeners, although they may have similar toxicological modes of action, have different toxicities to ecological receptors. Thus, the total PCDD or PCDF concentrations in a sample comprising a complex mixture of congeners may reveal relatively little about its toxicity. The most robust current approach to evaluating the potential risks posed by such mixtures is to estimate the toxicities of the congeners relative to that of 2,3,7,8-TCDD (the most well-studied and generally the most toxic of the dioxin congeners). To accomplish this, the concentration of each congener is converted to the equivalent 2,3,7,8-TCDD concentration (the TCDD-Equivalent or TCDD-EQ) using toxicity equivalence factors (TEFs). TEFs are the ratios of the toxicities of the congeners relative to that of 2,3,7,8-TCDD. In this ERA, the TEFs developed by the World Health Organization (Van den Berg *et al*., 1998) are used. Because these compounds act through the same mechanism, their toxicity is generally additive in environmentally relevant mixtures (Van den Berg *et al*., 1998). Thus, the total TCDD-EQ exposure is estimated by summing the TCDD-EQs for all the PCDD and PCDF congeners and any other compounds that share the same mechanism of action. Thus, the final TCDD-EQ value is a measure of the total toxicity of the mixture relative to 2,3,7,8-TCDD, and can be compared with toxicity reference values for that congener.

3.3 Conceptual Model

The purpose of conceptual models in ERA is to describe the relationships among environmental media, contaminants, and potentially exposed organisms, and to trace the pathways through which the ecological receptors may be exposed to the contaminant(s). By doing so, the conceptual model informs and directs the risk analysis. The Tittabawassee River and its floodplain are hydrologically linked in the assessment area (see Section 3-1 above), and contaminants flow between the riverine and terrestrial environments. For the sake of clarity, however, only the terrestrial contaminant-media-receptor interrelationships, are shown in the conceptual model in Figure 3-3.

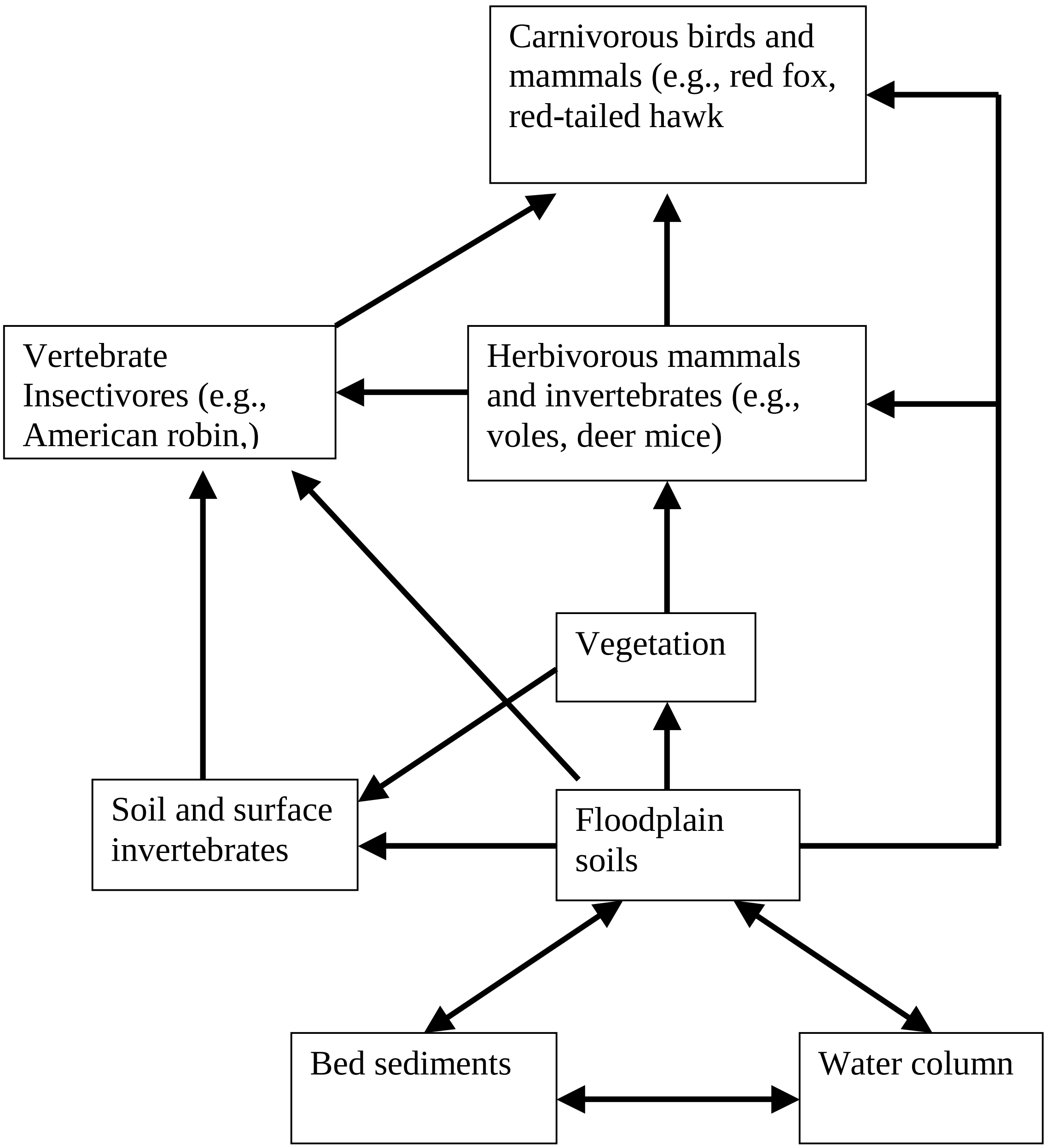


Figure 3-3. Conceptual model of contaminant transport into and through the terrestrial food chains of the Tittabawassee River floodplain

In the Tittabawassee River floodplain most of the PCDDs and PCDFs will be in the soils. From the soils they may be passed to subsurface and surface invertebrates by direct contact and through ingestion, and thence to insectivorous organisms that consume these invertebrates (Figure 3-3). Figure 3-3 also shows that herbivorous rodents may accumulate PCDDs and PCDFs through their diets, and that predators consuming these rodents may thereby be exposed. Insectivores and predators may also accumulate PCDDs and PCDFs by direct incidental ingestion of contaminated soil. Drinking contaminated surface waters also constitutes a theoretical exposure pathway to insectivores and predators. However, given their hydrophobic nature, the concentrations of PCDDs/PCDFs in the surface waters are likely to be so low as not to constitute a significant exposure route. Also, given their diets, the insectivores and predators will obtain a significant component, if not all, of their water needs through their prey. For these reasons, the water – biota exposure pathway is not considered in this ERA.

3.4 Assessment Endpoints

Assessment endpoints have been chosen for this ERA based on the known sensitivities of organisms at different levels in food chains and life stages to PCDDs and PCDFs (predators and early life stages being most vulnerable).

The assessment endpoints for this ERA are:

- Protection of avian carnivore and insectivore reproduction
- Protection of mammalian carnivore and insectivore reproduction

Because these endpoints represent protection of the species and life stages most vulnerable to PCDDs and PCDFs within the assessment area, they are likely to be protective of the other, less vulnerable, exposed ecological resources.

Assessment endpoints are general, non-quantitative statements about the resources that are to be protected through the ERA process. They do not provide quantitative targets or criteria on which the ERA can be based. However, they are important in that they provide focus for the ERA and provide the basis from which quantitative *measurement endpoints* can be established. These measurement endpoints are described in Sections 4.1 and 4.2 of this ERA.

3.5 Assessment Species

Reflecting the trophic and life-stage sensitivity to PCHs, discussed above in Section 3.2.1, this ERA focuses on evaluating risks to the reproductive performance of six wildlife species representing avian and mammalian insectivores and carnivorous predators. All occur in the Tittabawassee River floodplain. This section identifies these species and briefly describes their diets and distribution relative to the assessment area. Greater details on their diets, food and soil intake rates, and exposures to PCDD/PCDFs are given in Section 4.2.

Avian Carnivores

Two species were selected for analysis: the red-tailed hawk and the American kestrel (*Falco sparverius*). During the breeding season, red-tailed hawk diet comprises mainly small mammals and birds (Preston and Beane, 1993). The breeding season diet of the American kestrel comprises insects and small mammals and birds (Smallwood and Bird, 2002), with the latter two contributing the majority of the mass of the diet.

The breeding ranges of both species include the assessment area (Brewer *et al*., 1991; Price *et al*., 1995; Preston and Beane, 1993; Smallwood and Bird, 2002). Suitable wooded and agricultural habitat for both species is present in the assessment area and both species were seen on site visits in 2003 (*pers obs*). The red-tailed hawk is recorded as common on the Shiawassee NWR during spring and summer (http://www.npsc.nbs.gov/resource/othrdata/chekbird/r3/SHIAWA.HTM).

Avian Insectivores

Two avian insectivores were selected for this ERA: the American robin and the American woodcock. During the breeding season the majority of the American robin diet comprises surface and subsurface invertebrates (Sallabanks and James, 1999), while that of the woodcock comprises mainly earthworms (Keppie and Whiting, 1994).

The breeding ranges of both species include the assessment area (Brewer *et al*., 1991; Price *et al*., 1995; Sallabanks and James, 1999; Keppie and Whiting, 1994). Suitable wooded and agricultural habitat for both species is present in the assessment area and American robins were abundant on site visits in 2003 (*pers obs*). Both species are recorded from the Shiawassee NWR during spring and summer, with the American robin classified as abundant (http://www.npsc.nbs.gov/resource/othrdata/chekbird/r3/SHIAWA.HTM).

Mammalian Carnivore

The red fox (*Vulpes vulpes*) is the mammalian carnivore in this ERA. Its North American distribution includes the assessment area (Chapman and Feldhamer, 1982) and it is recorded from the Shiawassee NWR http://midwest.fws.gov/Shiawassee/PDF/refugewildlife.pdf). The diets of red foxes in the Midwest typically comprise carrion, small mammals, and small and medium sized birds (reported in U.S. EPA, 1993b).

Mammalian Insectivore

The short-tailed shrew (*Blarina brevicauda*) is the mammalian insectivore included in this ERA. Its North American distribution includes the assessment area (Chapman and Feldhamer, 1982) and its main habitats (woodland and agricultural land) are well represented on the Tittabawassee River floodplain. It has been recorded from the Shiawassee NWR (http://midwest.fws.gov/Shiawassee/PDF/refugewildlife.pdf). The diet of the short-tailed shrew comprises mainly earthworms and surface invertebrates (U.S. EPA, 1993b).

All of the above species are included in U.S. EPA's Exposure Factors Handbook (U.S. EPA, 1993b) where details of their demographics, diets, and ecologies are given.

4. ANALYSIS

This section describes the approaches followed and the values obtained in the determination of two important components of ecological risk evaluation:

- Estimating the exposure of the target organisms to PCDDs and PCDFs

- Developing stressor response relationships. In this ERA this translates into identifying the exposures of the receptors to the contaminants that are likely to be associated with toxicological responses (henceforward referred to as toxicity reference values or "TRVs"). The resulting values are the measurement endpoints described in Section 3.4.

Each of these stages is discussed separately for each receptor.

4.1 Exposures of Ecological Receptors

The exposure (i.e., daily dose) to any ecological receptor may be a function of a number of parameters including:

- Composition of the diet
- Contaminant residues in the prey
- Food intake rate
- Water contamination level
- Water intake rate
- Soil contamination level
- Soil ingestion rate
- Body mass of the receptor
- Proportion of the diet originating in the contaminated area

These components are addressed below for the six receptors in this ERA.

4.1.1 Diets, body masses, and food intake rates of receptors

The body masses, diets, and food intake rates for the six receptors were compiled from the scientific literature. They are described below and the resulting values used in this ERA are summarized in Table 4-1.

Table 4-1. Diets (by volume or biomass), body masses, and food intake rates of ecological receptors.

Species	**Diet**	**Body Mass (g)**	**Food Intake (g/d, ww)**
Red-tailed hawk	75% small mammals 25% birds	1,224	150
American kestrel	20% invertebrates 50% small mammals 30% birds	130	40
American woodcock	90% earthworms 10% other invertebrates	220	200
American robin	95% invertebrates 5% plant material	81	110
Red fox	50% small mammals 40% carrion 10% birds	4,100	400
Short-tailed shrew	50% surface invertebrates 30% earthworms 20% plant material	17.5	10

4.1.1.1 Red-tailed hawk

Diet

Diets of red-tailed hawks have been reported during spring (the period of oogenesis and laying) in two studies: In Wisconsin (Petersen, 1979, reported in Preston and Beane, 1993), the biomass of the diet comprised 59% small- to medium-sized mammals (voles and mice to rabbits) and 33% small- to medium-sized birds, with ring-necked pheasants (*Phasianus colchicus*) being major prey items. In Oregon, 78.5% of the spring diet was small mammals, 8.5% small birds, and 13.1% snakes (Janes, 1984). During summer, red-tailed hawks apparently rely more heavily on small- to medium-sized mammals: 73.7% of the diet in Alberta (Adamcik *et al*., 1979), and 94.2% of the diet in California (Fitch *et al*., 1946).

In this ERA it has been assumed, based on the above results, that the spring and summer diets of red-tailed hawks nesting on the Tittabawassee River floodplain will approximate 75% small mammals and 25% small- to medium-sized birds by volume or biomass.

Body mass

The average body mass of adult female red-tailed hawks in Michigan and Pennsylvania was 1.224 kg (Craighead and Craighead, 1956, reported in U.S. EPA, 1993b). In Ohio, Springer and Osborne (1983, reported in U.S. EPA, 1993b) report an average adult female body mass of 1.235 kg. In this ecological risk assessment, an average female body mass of 1.224 kg is assumed.

Food consumption

Craighead and Craighead (1956) estimated that during the summer months the daily food consumption of female red-tailed hawks was about 7% of their body weight. With an assumed body weight of 1,224 kg this equals 86 g/d (ww).

Using the non-passerine energetics equation 3-37 in U.S. EPA (1993b) (based on data in Nagy, 1987), the calculated daily energy requirement of a red-tailed hawk is:

$$1.146 \times 1224^{0.749} = 235 \text{ kcal/d}$$

If it is assumed that the energy content of the mammals and small birds eaten by red-tailed hawks approximate 1.8 kcal/g, ww, (Table 4-1 in U.S. EPA, 1993b), and that the digestive assimilation efficiency is 78% (Table 4-3 in U.S. EPA, 1993b), the daily food requirement is (235/1.8)x100/78 = 167g/d (ww).

Based on these two estimates, we assume that the daily food requirements for female red-tailed hawk during the energetically demanding time of oogenesis and laying is 150 g/d, wet weight (closer to the upper of the two estimates).

4.1.1.2 American kestrel

Diet

No data were found for the diet of American kestrels during spring in the northern or Great Lakes States. However, Sherrod (1978, reported in Smallwood and Bird, 2002) summarize the main year-round composition in North America as 74% invertebrates, 16% small mammals, and 9% small birds (by frequency in the diet). Because of the differences in masses among these dietary items, it is assumed that the dietary biomasses approximate 50% small mammals, 30% small birds, and 20% invertebrates (this assumes that invertebrates are, on average, one tenth the mass of small mammals and birds). These are the diet proportions used in this ERA.

Body mass

Average body masses for female kestrels have been reported as 120 g (Dunning, 1993, reported in Smallwood and Bird, 2002), 124 g during laying in Utah (Gessaman and Haggas, 1987, cited in U.S. EPA, 1993b), and 139.5 g during summer in New Jersey (Smallwood and Bird, 2002). In this ecological risk assessment, 130 g is assumed to be the average female body mass during oogenesis and egg laying on the Tittabawassee River floodplain.

Food consumption

The daily food ingestion rate of a captive male American kestrel was 0.31 g/g (bw)/d (Barret and MacKey, 1975). At an assumed body weight of 130 g, this equals 40.3 g/d (ww). Craighead and Craighead (1956) estimated a summer daily ingestion rate of American kestrels in Michigan of 27 g/d (ww). Gessaman and Haggas (1987) estimated that energy requirement of a free-living female American kestrel during laying was 414.4 kcal/kg/d. At an assumed body weight of 130 g, this translates into a daily requirement of 53.8 kcal/d. Assuming that the energy content of the small mammal, bird, and invertebrate prey is 1.7 kcal/g (Table 4-1 in U.S. EPA, 1993b), the daily food requirement for breeding female kestrels is 31.6 g/d (ww), similar to the values given above. In this ecological risk assessment it is assumed that the daily food ingestion rate of a female American kestrel during the energetically demanding time of oogenesis and laying is 40 g/d (ww).

4.1.1.3 American woodcock

Diet

The diet of the American woodcock comprises mainly invertebrates, with earthworms being the most heavily utilized prey taxon (review in Keppie and Whiting, 1994). The proportion of earthworms in the diet may vary seasonally, becoming less important during the summer months when earthworms burrow deeper and aestivate, and higher during fall through spring. Most of the available stomach contents data (reviewed in U.S. EPA, 1993b) are from the summer months when earthworms comprise between 58% and 68% of the stomach contents by volume (Krohn, 1970 and Sperry, 1940, respectively). One study (Stribling and Doerr, 1985 – reported in U.S. EPA, 1993b) reported that during the winter in North Carolina earthworms comprised more than 99% by volume of the items in the digestive tract.

In the northern U.S., American woodcocks begin breeding in late winter and early spring, with eggs being laid as early as late March (Keppie and Whiting, 1994). Thus, female woodcocks may be undergoing oogenesis from mid-March onwards. Because of this early breeding, it has been assumed that the winter diet composition is more representative of breeding season conditions. Thus, composition of the diet during egg formation and laying is assumed to be 90% by volume or biomass earthworms and the remainder other invertebrate species.

Body mass

During the breeding season in Massachusetts and Maine, average body masses of female woodcock were 229.6 g and 186.6 g, respectively (reported in Keppie and Whiting, 1994). Throughout the species' range, an average mass of 218 g has been reported for females (Nelson and Martin, 1953, reported in U.S. EPA, 1993b). In this ecological risk assessment, 220 g is assumed to be the average female body mass during oogenesis and egg laying on the Tittabawassee River floodplain.

Food consumption

In captivity with unlimited access to earthworms, American woodcocks consumed an average of 121 g/d (ww), (range: 18-208 g/d). A bird with a diet limited to only 61 g/d (ww) lost 30% of its body mass over 12 days, indicating that this intake level was not sufficient to maintain the condition of the bird, far less partition resources into egg formation (Liscinsky, 1972, reported in Keppie and Whiting, 1994). Daily intake requirements for female American woodcocks during the breeding season have been estimated (Keppie and Whiting, 1994) to vary from 24.0 to 41.4 g/d (dw). Assuming that earthworms are 84% water (Table 4-1, U.S. EPA, 1993b), the daily ingestion requirement is between 150 and 259 g (ww). In this ecological risk assessment it is assumed that the daily food ingestion rate for female American woodcocks during the energetically demanding time of oogenesis and laying is 200 g/d (ww).

4.1.1.4 American robin

Diet

A number of studies have quantified diets of adult American robins throughout the year, and of nestlings. The most reliable of these are based on analyses of stomach contents (rather than pellet or faecal analyses that may bias the results because of differences in the digestibility and detectability of the various taxa consumed [Galbraith, 1989; Galbraith *et al.*, 1993]). Wheelwright (1986) found that during the spring the diet of adult robins in the central U.S. comprised mainly invertebrates (92% by volume of the stomach contents). This proportion fell to 59% during the summer months and 24% and 27% during the fall and winter months, respectively. The remainder of the stomach contents in all four seasons comprised mainly fruits and seeds.

It is interesting that the Wheelwright (1986) results do not include earthworms among the most highly represented invertebrate taxa, whereas they comprised 15% by wet weight of the contents of adults robins during summer in New York State (Howell, 1942. This could be due to the fact that, in comparison to invertebrates with chitinous exoskeletons, earthworms are digested quickly, after which they become difficult to detect unless their highly inconspicuous setae can be found (Galbraith, 1989). Therefore, it could be that the Wheelwright (1986) analysis of stomach contents underestimates the proportions of earthworms in the diet of robins in spring.

In this ERA it has been assumed that the diet of adult robins during the spring comprises approximately 95% invertebrates and the remainder as fruits, seeds, or vegetation. This is comparable to the results that are reported in U.S. EPA (1993b), which are also based on the Wheelwright (1986) and Howell (1942) publications.

Body mass

In Pennsylvania, male and female American robins averaged 77.3 g year-round, while in New York State females during winter averaged 83.6 g (Sallabanks and James, 1999). Females during the breeding season in New York State averaged 80.6 g (reported in U.S. EPA, 1993b). In this ecological risk assessment, the average female body mass during oogenesis and egg laying in the Tittabawassee River floodplain is assumed to be 81 g.

Food consumption

Assuming a body weight of 81 g, and intake rates of free-living American robins reported in U.S. EPA (1993b), American robins ingest between 73 and 123 g/d (ww).

Using the passerine energetics equation 3-36 in U.S. EPA (1993b), the estimated daily energy requirement for American robins is:

$$2.123 \text{ x } (81^{0.749}) = 57.1 \text{ kcal/d.}$$

If it is then assumed that during the breeding season the diet comprises earthworms and other invertebrates, that the energy content of this diet is 1.0 kcal/g, ww, (Table 4-1, U.S. EPA 1993b), and that the assimilation efficiency of American robins is 72% (Table 4-3 in U.S. EPA (1993b)), the daily food ingestion rate is:

$$(57.1/1.0) \text{ x } 100/72 = 79 \text{ g/d (ww)}$$

Thus the range of estimated food intakes is from 73 – 123 g/d. In this ERA it is assumed that the daily food intake rate for female American robins during the energetically demanding period of oogenesis and laying is 110 g/d (ww) (closer to the upper part of the ranges given above).

4.1.1.5 Red fox

Diet

The red fox is an opportunistic predator and scavenger and its diet in any one area and season largely reflects what is available, most energetically profitable, and easiest to capture (Chapman and Feldhamer, 1982). During the mating and embryogenesis months of January- March [phenological data from Illinois, New York and Iowa in Chapman and Feldhamer (1982)] the majority of the diet of red foxes in Michigan comprised mainly deer carrion, with rabbits and mice also being taken (Schofield, 1960, in Chapman and Feldhamer, 1982); In Illinois, the diet comprised 65% small and medium sized mammals, 9% birds and 26% plants (Knable, 1974, reported in U.S. EPA, 1993b). In Missouri the diet comprised 69% mammals, 7.4% carrion, 15% birds, and 2% plant materials (Korschgen, 1959, reported in U.S. EPA, 1993b).

Based on the above data, it is assumed in this ERA that the diet (by volume or biomass) of the red fox in the Tittabawassee River floodplain during embryogenesis and rearing comprises 50% small mammals, 10% birds, and 40% carrion (equidistant between the Michigan and Missouri carrion representations).

Body mass

Storm *et al*., 1976 (cited in U.S. EPA, 1993b) report average red fox female body masses during the spring ranging between 3.3 and 4.7 kg, with an average of 4.1 kg. This last body mass is assumed in this ecological risk assessment to be the average body mass on the Tittabawassee River floodplain.

Food consumption

Sargeant (1978, reported in U.S. EPA, 1993b) found that the food intake rates of a captive female red fox before and after whelping were 307 – 570 g/d, respectively. Using the energetics equation 3-11 in U.S. EPA (1993b) the daily caloric intake rate for a 4.1 kg red fox is 640 kcal. At 1.8 kcal/g metabolizable energy in the diet (Table 4-1, U.S. EPA, 1993b), the estimated food intake rate becomes 640/1.8 = 355 g/d (ww). With an assimilation efficiency of 84% (Table 4-3, U.S, EPA, 1993b), the final value for the daily intake rate is 422 g/d (ww). In this ERA it is assumed that the daily food ingestion rate for the red fox is 400 g/d.

4.1.1.6 Short-tailed shrew

Diet

While short-tailed shrews may breed year-round, mating and parturition in Indiana and New York State peak between April and June (Chapman and Feldhamer, 1982). During the summer in New York State and the eastern U.S. the main components of the diet (based on % volume of the stomach contents) comprised 31% earthworms, 27% slugs and snails, and 20% other invertebrates (Whittaker and Ferraro, 1963, reported in Chapman and Feldhamer, 1982). In this ERA it is assumed that the diet of short-tailed shrews in the Tittabawassee River floodplain comprises 30% earthworms and 50% other surface-dwelling invertebrates. The remaining 20% is assumed to be plant and fungal material.

Body mass

Short-tailed shrew summer and fall body masses ranging between 12.5 and 22.5 g are reported for Pennsylvania by Guilday (1957), cited in U.S. EPA (1993b). The midpoint of this range (17.5 g) is assumed to be the average female body mass during embryogenesis and parturition on the Tittabawassee River floodplain.

Food consumption

The food intake rates of short-tailed shrews in the laboratory varied between 0.49 and 0.62 g/g(bw)/day (Barrett and Stuek, 1976, and Morrison *et al*., 1957, reported in U.S. EPA, 1993b). Therefore, assuming a body weight of 17.5, the estimated daily food intake rate for the short-tailed shrew is between 8.6 and 10.8 g/d. In this ERA it is assumed that the daily food intake rate is 10 g/d.

4.1.2 Receptor soil ingestion rates

4.1.2.1 Red-tailed hawk

The daily soil ingestion rate for red-tailed hawks is assumed to approximate that of red foxes (since they eat similar prey). Table 4-4 in U.S. EPA (1993b) gives this as 2.8% of the diet (dw). Assuming a daily food intake of 150 g/d (ww), and a dietary water composition of approximately 70% (Table 4-1, U.S.EPA, 1993b), an estimated 1.3 g (dw) of soil is ingested daily.

4.1.2.2 American kestrel

The daily soil ingestion rate of American kestrels is assumed to be similar to that of the red-tailed hawk at 2.8% of the diet (dw). Assuming 70% water in the diet (Table 4-1, U.S. EPA, 1993b), this translates into 0.4 g/d (dw) of soil.

4.1.2.3 American robin

The daily soil ingestion rate of American robins is assumed to be less than that of the more-strictly vermivorous American woodcock (10.4% of the diet (dw)), since the diet comprises a lower proportion of earthworms, but greater than that of vegetarian mammals and birds (2-3% of the diet). The value assumed in this ecological risk assessment is 5% of the diet (dw). Assuming a 70% water content in the diet (Table 4-1, U.S. EPA, 1993b), this translates into 1.6 g/d (dw) of soil.

4.1.2.4 American woodcock

The American woodcock soil intake rate is given in U.S. EPA (1993b) as 10.4% of the diet (dw), which, assuming an 84% water content in the diet (Table 4-1, U.S. EPA, 1993b), translates into a daily intake of 3.3 g (dw) of soil.

4.1.2.5 Red fox

The red fox soil ingestion rate is given in U.S. EPA (1993b) as 2.8% (dw) of the diet. Assuming a dietary intake rate of 400 g (ww)/d and a dietary water composition of 70% (from Table 4-1 in U.S. EPA, 1993b), this translates into a daily soil intake of 3.4 g (dw).

4.1.2.6 Short-tailed shrew

Approximating from the white-footed mouse and vole data in Table 4-4 of U.S. EPA (1993b), the daily soil intake rate for the short-tailed shrew is estimated to 2% (dw) of the diet. Assuming a dietary water content of approximately 70% (Table 4-1, U.S. EPA 1993b), this translates into a daily soil ingestion rate of 0.06 g (dw)/d.

4.1.3 Proportion of receptor diet from the assessment area

The proportion of its diet that any receptor would obtain from the assessment area is a function of the extent of its foraging range relative to that of the 9.6 square miles (2,500 ha, 6,144 acres) of the contaminated floodplain, and the quality of the floodplain habitat. Assuming equally attractive habitats outside of and within the contaminated area, any organism whose foraging range exceeded that of the contaminated area would obtain at least some of its food from outside the area.

In this ERA the home or foraging range sizes of the six receptor species have been reviewed in the scientific literature. The results are described below and summarized in Table 4-2.

4.1.3.1 Red-tailed hawk

The extents of red-tailed hawk breeding territories vary with habitat quality. However, in Wisconsin during the spring and summer home ranges of females ranged from 31 to 206

Table 4-2. Home range sizes of the ecological receptors.	
Species	**Home Range (ha)**
Red-tailed hawk	100
American kestrel	130
American woodcock	35
American robin	0.5
Red fox	500
Short-tailed shrew	0.3

ha (Petersen, 1979, reported in Preston and Beane, 1993). In this ecological risk assessment we assume that the spring and summer home ranges (i.e., foraging areas) of female red-tailed hawks approximate 100 ha in extent. Given the size of the floodplain under investigation, it is reasonable to assume that at least some of the red-tailed hawks that could nest there could obtain their entire diet from the floodplain, itself.

4.1.3.2 American kestrel

During summer in Michigan, Craighead and Craighead (1956) found that the territories of American kestrels ranged up to 215 ha with a mean of 131 ha. Bird and Palmer (1988) report typical nesting densities ranging from 0.11 to 1.74 pairs/km^2. This translates into home range sizes of 57 to 900 ha. (assuming that territories are exclusive and that all habitat is used). In this ecological risk assessment it is assumed that the foraging range of female American kestrels during breeding will approximate 130 ha. Thus, it is not unreasonable to assume that at least some of any American kestrels that might nest on the floodplain of the Tittabawassee River could obtain their entire diet from the floodplain, itself.

4.1.3.3 American woodcock

In Maine in summer, the home ranges of adult female American woodcocks averaged 42 ha (Sepik and Derleth, 1993, reported in Keppie and Whiting, 1994). During spring and summer in Wisconsin, the home ranges of adult American woodcocks averaged 4.5 ha and 32.4 ha, respectively (Gregg, 1984, reported in U.S. EPA, 1993b). For this ecological risk assessment, the late winter and early spring foraging ranges of female woodcocks in the Tittabawassee River floodplain are assumed to approximate 35 ha. Thus, many individual woodcocks could have their entire foraging range within the floodplain.

4.1.3.4 American robin

In Ontario, Canada, the foraging ranges of American robins when feeding nestlings and fledglings were 0.15 and 0.81 ha, respectively (Weatherhead and McRae, 1990 reported in U.S. EPA, 1993b). In New York State, breeding territories ranged between 0.11 and 0.21 ha (Howell, 1942, reported in Sallabanks and James, 1999). In Wisconsin, breeding territories ranged from 0.04 to 0.24 ha (Young, 1951, reported in Sallabanks and James, 1999). In this ecological risk assessment, it is assumed that breeding territories approximate foraging ranges. Thus, the foraging range of female American robins during oogenesis on the Tittabawassee River floodplain is assumed to be 0.5 ha. Therefore, many individual American robins could have their entire foraging ranges within the floodplain.

4.1.3.5 Red fox

In Wisconsin the home ranges of female red foxes ranged from 57 to 170 ha, averaging 96 ha. (Ables, 1969, reported in U.S. EPA 1993b). In spring in Minnesota, the home ranges of females ranged between 596 and 855 ha, and averaged 699 ha (Sargeant, 1972, reported in U.S. EPA, 1993b). Follman (1973), (cited in Chapman and Feldhamer, 1982) found that home range sizes of female red foxes contracted during the early breeding season. In this ecological risk assessment it is assumed that the foraging range of female red foxes during embryogenesis and parturition approximates 500 ha. Since the assessment area approximates 2,500 ha, red foxes could obtain their entire diet from within the assessment area.

4.1.3.6 Short-tailed shrew

In south Michigan, summer home ranges of short-tailed shrews ranged between <0.1 and 0.36 ha (Blair, 1940, cited in U.S. EPA, 1993b). In New York State, winter home ranges of up to 0.22 ha were recorded (Platt, 1976, cited in U.S. EPA, 1993b). In this ecological risk assessment, an average home range size of 0.3 ha is assumed. This implies that many of the short-tailed shrews living on the floodplain would obtain their entire diet from the floodplain, itself.

4.1.4 Consumption of contaminated water

Drinking contaminated surface waters also constitutes a theoretical exposure pathway to insectivores and predators in the assessment area. However, given their hydrophobic nature, the concentrations of PCDDs/PCDFs in the surface waters are likely to be so low as to not constitute a significant exposure route. Also, given their diets, the insectivores and predators will obtain a significant component, if not all, of their water needs through their prey. Accordingly, a water – biota exposure pathway is not considered in this ERA.

4.1.5 Uptake factors

In this screening-level ERA the PCDD/PCDF and TCDD-EQ concentrations in the invertebrate and vertebrate prey of the six receptors are estimated from known soil concentrations using uptake factors (UFs) obtained from the scientific literature. UFs are ratios established in empirical co-locational studies between contaminant concentrations in soil and organisms. Since different PCDD/PCDF congeners may differ in their propensities to bioaccumulate, UFs must be expressed on a congener-specific basis.

A review of the scientific literature found relevant empirically-derived UFs for 2,3,7,8-TCDD, 2,3,7,8-TCDF, and 2,3,4,7,8-PeCDF. These are presented in Table 4-3.

Table 4-3. UFs reported in the scientific literature for soil – invertebrates and vertebrates (soil dry weight and organism wet weight)[a]

	2,3,7,8-TCDD	2,3,7,8-TCDF	2,3,4,7,8-PeCDF	Authority
Earthworm	5			Reinecke and Nash, 1984
	0.6			Heida *et al*., 1986
			0.3/0.04[b]	Heida *et al*., 1986
	3.3			Thiel *et al*., 1989
	14.5			Martinucci *et al*., 1983
	10			Fanelli *et al*., 1982
	1.9/6.4[c]			Sample *et al*., 1998
Deer mouse	1.4	0.07		Thiel *et al*., 1989
Field mouse	1.3			Fanelli *et al*., 1982
Woodmouse/shrews	2.8	1.5	59/27[b]	Heida *et al*., 1986
Eastern bluebird	0.2	0.007		Thiel *et al*., 1988

[a]Where necessary, dry weight organism data converted to wet weights using water content conversion factors of 84% and 68% for earthworms and mammals, respectively (U.S. EPA, 1993b).
[b]values to left and right of slash indicate results from drier and wetter and less and more organic soils, respectively.
[C]values to left and right of slash are mean and upper 95% prediction level

From the results in Table 4-3 it is assumed that a reasonably protective UF for 2,3,7,8-TCDD to earthworms is 5 (approximately the midpoint of the various estimates). For 2,3,4,7,8-PeCDF the soil-earthworm UF is assumed to be 0.1. No soil-earthworm UF data were found for TCDF and 0.1 was assumed. For soil-small mammals, UFs of 2, 1, and 10 are assumed for TCDD, 2,3,7,8-TCDF, and 2,3,4,7,8-PCDF, respectively. Only one study was found that reported soil-body uptake of TCDD and TCDF in birds (Thiel *et al*., 1988). However, this study probably underestimates the UF since it was based on sampling eastern bluebirds during egg laying, a time when the birds were actively translocating body resources into their second clutches (much of their body burden may have already been transferred into their first clutches). In this ERA it assumed that a more representative soil-bird TCDD UF is 1, half that of small mammals. It is also assumed that the soil-bird UFs for TCDF and PCDF are also approximately half those of small mammals. The resulting UFs that are used in this ERA are summarized in Table 4-4.

Table 4-4. Soil- organism UFs used in this ecological risk assessment			
	2,3,7,8-TCDD	**2,3,7,8-TCDF**	**2,3,4,7,8-PeCDF**
Earthworm	5	0.1	0.1
Rodents/shrews	2	1	10
Birds	1	0.5	5

4.1.6 Floodplain soil and estimated biota concentrations of PCDD, PCDF, and TCDD-EQ

Table 4-5 shows the mean surface soil (0-3 inches depth) congener concentrations measured by MDEQ during 2002 and 2003 in the Tittabawassee River floodplain upriver and downriver of the City of Midland. These data show that all the congener concentrations downriver of Midland are much higher than those from upriver (by factors of 2 and 3 orders of magnitude in the cases of 2,3,4,7,8-PeCDF and 2,3,7,8-TCDF, respectively). Table 4-5 also uses the UFs from Table 4-4 to estimate congener tissue concentrations in invertebrates, small mammals, and birds in the floodplain upriver and downriver of Midland. Again, in all cases the estimated mean congener concentrations downriver of Midland are higher (by up to 3 orders of magnitude) than those from upriver. UFs were not found for most of the congeners listed in Table 4-5. It was assumed that, except for the UFs listed in Table 4-4, the unknown UFs approximate 0.1. This may underestimate the contaminant uptake of the receptors, however analyses described below show that the vast majority of the TCDD-EQ exposure to the receptors is due to the three congeners for which UFs were available so it is not expected that this would be an important bias.

Table 4-6 shows the estimated mean tissue concentrations in Table 4-5 converted to TCDD-EQ using WHO avian TEFS. Table 4-7 translates the data in Table 4-6 into estimated mean total TCDD-EQ in tissues of invertebrates, small mammals, and birds in the Tittabawassee River floodplain upriver and downriver of the City of Midland. Table 4-8 uses the data in the previous three tables to estimate the percent contributions of three individual congeners (2,3,7,8-TCDD, 2,3,7,8-TCDF, and 2,3,4,7,8-PeCDF) to the total estimated TCDD-EQ in invertebrate, small mammal, and bird tissues downriver of Midland. Tables 4-9 and 4-10 are similar to Tables 4-6 and 4-7 except that WHO mammalian TEFS are used to estimate TCDD-EQ.

Table 4-5. Mean PCDD and PCDF congener concentrations (pg/g dw) in floodplain soils (0-3 inches depth) upriver and downriver of the City of Midland, uptake factors, and estimated mean tissue concentrations in invertebrates, small mammals, and birds (pg/g ww)

Congener	Reach	N	Mean Soil Conc.	Invertebrate UF	Estimated Mean Invertebrate Tissue Conc.	Small Mammal UF	Estimated Mean Small Mammal Tissue Conc.	Bird UF	Estimated Mean Bird Tissue Conc.
2,3,7,8-TCDD	Upriver	12	2.1	5	10.5	2	4.2	1	2.1
	Downriver	53	6.1		30.5		12.2		6.1
1,2,3,7,8-PCDD	Upriver	12	1.5	0.1*	0.1	0.1*	0.1	0.1*	0.1
	Downriver	53	6.9		0.7		0.7		0.7
1,2,3,4,7,8-HxCDD	Upriver	12	1.1	0.1*	0.1	0.1*	0.1	0.1*	0.1
	Downriver	53	19.5		1.9		1.9		1.9
1,2,3,6,7,8-HxCDD	Upriver	12	2.9	0.1*	0.3	0.1*	0.3	0.1*	0.3
	Downriver	53	27.4		2.7		2.7		2.7
1,2,3,7,8,9-HxCDD	Upriver	12	2.8	0.1*	0.3	0.1*	0.3	0.1*	0.3
	Downriver	53	12.6		1.3		1.3		1.3
1,2,3,4,6,7,8-HpCDD	Upriver	12	41.2	0.1*	4.1	0.1*	4.1	0.1*	4.1
	Downriver	53	546		55		55		55
OCDD	Upriver	12	321	0.1*	32	0.1*	32	0.1*	32
	Downriver	53	5,210		521		521		521
2,3,7,8-TCDF	Upriver	12	5.2	0.1	0.5	1	5.2	0.5	2.6
	Downriver	53	2,388		239		2,388		1,194
1,2,3,7,8-PeCDF	Upriver	12	4.3	0.1*	0.4	0.1*	0.4	0.1*	0.4
	Downriver	53	1,334		133		133		133
2,3,4,7,8-PeCDF	Upriver	12	2.5	0.1	0.2	10	25	5	12.5
	Downriver	53	962		96		9,620		4810
1,2,3,4,7,8-HxCDF	Upriver	12	8.3	0.1*	0.8	0.1*	0.8	0.1*	0.8
	Downriver	53	930		93		93		93

Table 4-5 continued									
Congener	**Reach**	**N**	**Mean Soil Conc.**	**Invertebrate UF**	**Estimated Mean Invertebrate Tissue Conc.**	**Small Mammal UF**	**Estimated Mean Small Mammal Tissue Conc.**	**Bird UF**	**Estimated Mean Bird Tissue Conc.**
1,2,3,6,7,8-HxCDF	Upriver	12	2.2	0.1*	0.2	0.1*	0.2	0.1*	0.2
	Downriver	53	167		17		17		17
2,3,4,6,7,8-HxCDF	Upriver	12	1.9	0.1*	0.2	0.1*	0.2	0.1*	0.2
	Downriver	53	106		10.6		10.6		10.6
1,2,3,7,8,9-HxCDF	Upriver	12	0.8	0.1*	0.1	0.1*	0.1	0.1*	0.1
	Downriver	53	15.7		1.6		1.6		1.6
1,2,3,4,6,7,8-HpCDF	Upriver	12	21.5	0.1*	2.1	0.1*	2.1	0.1*	2.1
	Downriver	53	1,082		108		108		108
1,2,3,4,7,8,9-HpCDF	Upriver	12	2.1	0.1*	0.2	0.1*	0.2	0.1*	0.2
	Downriver	53	73		7.3		7.3		7.3
OCDF	Upriver	12	28.5	0.1*	2.8	0.1*	2.8	0.1*	2.8
	Downriver	53	1,342		134		134		134

*Assumed uptake factors

Table 4-6. Estimated mean TCDD-EQ concentrations (pg/g, ww) in tissues of invertebrates, small mammals, and birds in the Tittabawassee River floodplain upriver and downriver of the City of Midland. TCDD-EQ estimated using WHO avian TEFs

Congener	Reach	N	Estimated Mean Invertebrate Tissue Conc.	TEF	Estimated Mean TCDD-EQ	Estimated Mean Small Mammal Tissue Conc.	TEF	Estimated Mean TCDD-EQ	Estimated Mean Bird Tissue Conc.	TEF	Estimated Mean TCDD-EQ
2,3,7,8-TCDD	Upriver	12	10.5	1	10.5	4.2	1	4.2	2.1	1	2.1
	Downriver	53	30.5		30.5	12.2		12.2	6.1		6.1
1,2,3,7,8-PCDD	Upriver	12	0.1	1	0.1	0.1	1	0.1	0.1	1	0.1
	Downriver	53	0.7		0.7	0.7		0.7	0.7		0.7
1,2,3,4,7,8-HxCDD	Upriver	12	0.1	0.05	0.005	0.1	0.05	0.005	0.1	0.05	0.005
	Downriver	53	1.9		0.09	1.9		0.09	1.9		0.09
1,2,3,6,7,8-HxCDD	Upriver	12	0.3	0.01	0.003	0.3	0.01	0.003	0.3	0.01	0.003
	Downriver	53	2.7		0.03	2.7		0.03	2.7		0.03
1,2,3,7,8,9-HxCDD	Upriver	12	0.3	0.1	0.03	0.3	0.1	0.03	0.3	0.1	0.03
	Downriver	53	1.3		0.13	1.3		0.13	1.3		0.13
1,2,3,4,6,7,8-HpCDD	Upriver	12	4.1	<0.001	0.004	4.1	<0.001	0.004	4.1	<0.001	0.004
	Downriver	53	55		0.05	55		0.05	55		0.05
OCDD	Upriver	12	32	0.0001	0.003	32	0.0001	0.003	32	0.0001	0.003
	Downriver	53	521		0.05	521		0.05	521		0.05
2,3,7,8-TCDF	Upriver	12	0.5	1	0.5	5.2	1	5.2	2.6	1	2.6
	Downriver	53	239		239	2,388		2,388	1,194		1,194
1,2,3,7,8-PeCDF	Upriver	12	0.4	0.1	0.04	0.4	0.1	0.04	0.4	0.1	0.04
	Downriver	53	133		13.3	133		13.3	133		13.3
2,3,4,7,8-PeCDF	Upriver	12	0.2	1	0.2	25	1	25	12.5	1	12.5
	Downriver	53	96		96	9,620		9,620	4,810		4,810

Table 4-6 continued.

Congener	Reach	N	Estimated Mean Invertebrate Tissue Conc.	TEF	Estimated Mean TCDD-EQ	Estimated Mean Small Mammal Tissue Conc.	TEF	Estimated Mean TCDD-EQ	Estimated Mean Bird Tissue Conc.	TEF	Estimated Mean TCDD-EQ
1,2,3,4,7,8-HxCDF	Upriver	12	0.8	0.1	0.08	0.8	0.1	0.08	0.8	0.1	0.08
	Downriver	53	93		9.3	93		9.3	93		9.3
1,2,3,6,7,8-HxCDF	Upriver	12	0.2	0.1	0.02	0.2	0.1	0.02	0.2	0.1	0.02
	Downriver	53	17		1.7	17		1.7	17		1.7
2,3,4,6,7,8-HxCDF	Upriver	12	0.2	0.1	0.02	0.2	0.1	0.02	0.2	0.1	0.02
	Downriver	53	10.6		1.1	10.6		1.1	10.6		1.1
1,2,3,7,8,9-HxCDF	Upriver	12	0.1	0.1	0.01	0.1	0.1	0.01	0.1	0.1	0.01
	Downriver	53	1.6		0.2	1.6		0.2	1.6		0.2
1,2,3,4,6,7,8-HpCDF	Upriver	12	2.1	0.01	0.02	2.1	0.01	0.02	2.1	0.01	0.02
	Downriver	53	108		1.1	108		1.1	108		1.1
1,2,3,4,7,8,9-HpCDF	Upriver	12	0.2	0.01	0.002	0.2	0.01	0.002	0.2	0.01	0.002
	Downriver	53	7.3		0.07	7.3		0.07	7.3		0.07
OCDF	Upriver	12	2.8	0.0001	0.0003	2.8	0.0001	0.0003	2.8	0.0001	0.0003
	Downriver	53	134		0.01	134		0.01	134		0.01

Table 4-7. Estimated mean total TCDD-EQ (pg/g, ww) in tissues of invertebrates, small mammals, and birds in the floodplain upriver and downriver of the City of Midland. TCDD-EQ estimated using WHO avian TEFs.			
	Mean Total Invertebrate TCDD-EQ	**Mean Total Small Mammal TCDD-EQ**	**Mean Total Bird TCDD-EQ**
Upriver	11.5	34.7	17.5
Downriver	393	12,048	6,038

Table 4-8. Percent contributions of congeners to total TCDD-EQ in tissues of invertebrates, small mammals, and birds.			
	Invertebrates	**Small Mammals**	**Birds**
2,3,7,8-TCDD	8%	0.1%	0.1%
2,3,7,8-TCDF	61%	19.8%	19.7
2,3,4,7,8-PeCDF	24%	80%	79%

Tables 4-6 through 4-10 show that the estimated mean TCDD-EQ concentrations in invertebrate, small mammal, and bird tissues are typically one to three orders of magnitude higher upriver than downriver of midland. Moreover, Table 4-8 shows that most of the total estimated TCDD-EQ in invertebrate and small mammal tissue downriver of Midland is due mainly to two congeners: 2,3,7,8-TCDF and 2,3,4,7,8-PeCDF.

Table 4-9. Estimated TCDD-EQ concentrations (pg/g, ww) in tissues of invertebrates, small mammals and birds in the Tittabawassee River floodplain upriver and downriver of the City of Midland. TCDD-EQ estimated using WHO mammalian TEFs											
Congener	**Reach**	**N**	**Estimated Invertebrate Tissue Conc.**	**TEF**	**Estimated TCDD-EQ**	**Estimated Rodent/Shrew Tissue Conc.**	**TEF**	**Estimated TCDD-EQ**	**Estimated Bird Tissue Conc.**	**TEF**	**Estimated TCDD-EQ**
2,3,7,8-TCDD	Upriver	12	10.5	1	10.5	4.2	1	4.2	2.1	1	2.1
	Downriver	53	30.5		30.5	12.2		12.2	6.1		6.1
1,2,3,7,8-PCDD	Upriver	12	0.1	1	0.1	0.1	1	0.1	0.1	1	0.1
	Downriver	53	0.7		0.7	0.7		0.7	0.7		0.7
1,2,3,4,7,8-HxCDD	Upriver	12	0.1	0.1	0.01	0.1	0.1	0.01	0.1	0.1	0.01
	Downriver	53	1.9		0.19	1.9		0.19	1.9		0.19
1,2,3,6,7,8-HxCDD	Upriver	12	0.3	0.1	0.03	0.3	0.1	0.03	0.3	0.1	0.03
	Downriver	53	2.7		0.3	2.7		0.3	2.7		0.3
1,2,3,7,8,9-HxCDD	Upriver	12	0.3	0.1	0.03	0.3	0.1	0.03	0.3	0.1	0.03
	Downriver	53	1.3		0.13	1.3		0.13	1.3		0.13
1,2,3,4,6,7,8-HpCDD	Upriver	12	4.1	0.001	0.004	4.1	0.001	0.004	4.1	0.001	0.004
	Downriver	53	55		0.05	55		0.05	55		0.05
OCDD	Upriver	12	32	0.0001	0.003	32	0.0001	0.003	32	0.0001	0.003
	Downriver	53	521		0.05	521		0.05	521		0.05
2,3,7,8-TCDF	Upriver	12	0.5	0.1	0.05	5.2	0.1	0.5	2.6	0.1	0.3
	Downriver	53	239		23.9	2,388		239	1,194		119
1,2,3,7,8-PeCDF	Upriver	12	0.4	0.05	0.02	0.4	0.05	0.02	0.4	0.05	0.02
	Downriver	53	133		6.6	133		6.6	133		6.6
2,3,4,7,8-PeCDF	Upriver	12	0.2	0.5	0.1	25	0.5	12.5	12.5	0.5	6.2
	Downriver	53	96		48	9,620		4,810	4,810		2,405
1,2,3,4,7,8-HxCDF	Upriver	12	0.8	0.1	0.08	0.8	0.1	0.08	0.8	0.1	0.08
	Downriver	53	93		9.3	93		9.3	93		9.3

Table 4-9 continued.											
Congener	**Reach**	**N**	**Estimated Invertebrate Tissue Conc.**	**TEF**	**Estimated TCDD-EQ**	**Estimated Rodent/Shrew Tissue Conc.**	**TEF**	**Estimated TCDD-EQ**	**Estimated Bird Tissue Conc.**	**TEF**	**Estimated TCDD-EQ**
1,2,3,6,7,8-HxCDF	Upriver	12	0.2	0.1	0.02	0.2	0.1	0.02	0.2	0.1	0.02
	Downriver	53	17		1.7	17		1.7	17		1.7
2,3,4,6,7,8-HxCDF	Upriver	12	0.2	0.1	0.02	0.2	0.1	0.02	0.2	0.1	0.02
	Downriver	53	10.6		1.1	10.6		1.1	10.6		1.1
1,2,3,7,8,9-HxCDF	Upriver	12	0.1	0.1	0.01	0.1	0.1	0.01	0.1	0.1	0.01
	Downriver	53	1.6		0.2	1.6		0.2	1.6		0.2
1,2,3,4,6,7,8-HpCDF	Upriver	12	2.1	0.01	0.02	2.1	0.01	0.02	2.1	0.01	0.02
	Downriver	53	108		1.1	108		1.1	108		1.1
1,2,3,4,7,8,9-HpCDF	Upriver	12	0.2	0.01	0.002	0.2	0.01	0.002	0.2	0.01	0.002
	Downriver	53	7.3		0.07	7.3		0.07	7.3		0.07
OCDF	Upriver	12	2.8	0.0001	0.0003	2.8	0.0001	0.0003	2.8	0.0001	0.0003
	Downriver	53	134		0.01	134		0.01	134		0.01

Table 4-10. Estimated mean total TCDD-EQ (pg/g, ww) in tissues of invertebrates, small mammals, and birds in the floodplain upriver and downriver of the City of Midland. TCDD-EQ estimated using WHO mammalian TEFs.			
	Mean Total Invertebrate TCDD-EQ	**Mean Total Small Mammal TCDD-EQ**	**Mean Total Bird TCDD-EQ**
Upriver	11.0	17.5	8.9
Downriver	124	5,083	2,552

Table 4-11 shows the mean, maximum, median and 95% UCL on the meanTCDD-EQ concentrations for all congeners in floodplain soils (0-3 inches) downriver of Midland (MDEQ – unpublished data). These data are used in Section 4.1.7 of this ERA in the estimation of exposures to receptors from ingestion of soils, and in Section 5 to calculate alternative Hazard Indices (HI).

Table 4-11. Mean, median, maximum, and 95% UCL TCDD-EQ concentrations (pg/g, dw) in floodplain soils downriver of Midland.				
	Mean TCDD-EQ concentration	**Median TCDD-EQ concentration**	**Maximum TCDD-EQ concentration**	**95% UCL TCDD-EQ concentration**
WHO avian TEF	3,632	1,525	31,420	5,259
WHO mammalian TEF	945	446	7,421	1,332

4.1.7 TCDD-EQ exposures to receptors

In this section, the estimated food and soil intake rates (Sections 4.1.1 and 4.1.2, respectively) are combined with the TCDD-EQ estimated concentrations in the prey and soils (Tables 4-7, 4-10, and 4-11) to calculate the total daily doses of TCDD-EQ to the six receptor species. The daily dose to any one of the receptors from dietary and soil ingestion is described in the following equation:

$$DD = (C_f \times FI) + (C_s \times SI)$$

Where: DD is the daily dose of TCDD-EQ
C_f is the TCDD-EQ concentration in the diet
C_s is the TCDD-EQ concentration in the ingested soil
FI is the daily food ingestion rate
SI is the daily soil ingestion rate

Daily exposure doses to each of the four receptor species are calculated below:

4.1.7.1 Red-tailed hawk

The red-tailed hawk is potentially exposed to PCDDs and PCDFs in the Tittabawassee River floodplain through ingestion of small mammals (75% of the diet), birds (25% of the diet) and soils (1.3 g/d). The TCDD-EQ contributions from each of these sources are shown below.

Small mammals: [12,048 x (150 x 0.75)] = 1,355,400 pg/d TCDD-EQ

Birds: [6,038 x (150 x 0.25)] = 226,425 pg/d TCDD-EQ

Soil: 3,632 x 1.3 = 4,722 pg/d TCDD-EQ

The total daily TCDD-EQ intake is 1,586,547 pg

4.1.7.2 American kestrel

The American kestrel is potentially exposed to PCDDs and PCDFs in the Tittabawassee River floodplain through ingestion of invertebrates (20% of the diet), small mammals (50% of the diet), birds (30% of the diet) and soils (0.4 g/d). The TCDD-EQ contributions of each of these sources are shown below.

Invertebrates: 393 x (40 x 0.2) = 3,144 pg/d

Small mammals: 12,048 x (40 x 0.5) = 240,960 pg/d

Birds: 6,038 x (40 x 0.3) = 72,456 pg/d

Soil: 3,632 x 0.4 = 1453 pg/d

The total daily TCDD-EQ intake is 318,013 pg

4.1.7.3 American woodcock

The American woodcock is potentially exposed to PCDDs and PCDFs in the Tittabawassee River floodplain through ingestion of invertebrates (100% of the diet) and soils (3.3 g/d). The TCDD-EQ contributions of each of these sources are shown below.

Invertebrates: 393 x 200 = 78,600 pg/d

Soil: 3,632 x 3.3 = 11,986 pg/d

The total daily TCDD-EQ intake is 90,586 pg

4.1.7.4 American robin

The American robin is potentially exposed to PCDDs and PCDFs in the Tittabawassee River floodplain through ingestion of invertebrates (95% of the diet), plant material (5% of the diet) and soils (1.6 g/d). The plant contribution is likely to be extremely small and is not considered in this ERA. The TCDD-EQ contributions from the remaining two sources are shown below.

Invertebrates: 393 x (110 x 0.95) = 41,068 pg/d

Soil: 3,632 x 1.6 = 5,811 pg/d

The total daily TCDD-EQ intake is 46,879 pg

4.1.7.5 Red fox

The red fox is potentially exposed to PCDDs and PCDFs in the Tittabawassee River floodplain through ingestion of small mammals (50% of the diet), carrion (40% of the diet), birds (10% of the diet), and soils (3.4 g/d). Since much of the carrion consumed is likely to be small mammals and birds it is assumed that the TCDD-EQ in this food source is their average). The TCDD-EQ contributions from each of these sources are shown below.

Small mammals: 5,083 x (400 x 0.5) = 1,016,600 pg/d

Carrion: 3,817 x (400 x 0.4) = 610,720 pg/d

Birds: 2,552 x (400 x 0.1) = 102,080 pg/d

Soil: 3.4 x 945 = 3,213

The total daily TCDD-EQ intake is 1,732,613 pg

4.1.7.6 Short-tailed shrew

The short-tailed shrew is potentially exposed to PCDDs and PCDFs in the Tittabawassee River floodplain through ingestion of invertebrates (80% of the diet), plant material (20% of the diet), and soil (0.06 g/d). The plant contribution is likely to be relatively small and is not considered in this ERA. The TCDD-EQ contributions from each of the remaining sources are shown below.

Invertebrates: 124 x (10 x 0.8) = 992 pg/d

Soil: 0.06 x 945 = 56.7 pg/d

The total daily TCDD-EQ intake is 1,049 pg

4.2 Toxicity Reference Values

4.2.1 Insectivorous and Carnivorous Birds

The most sensitive stage of avian life-history to PCH toxicity is reproduction (Gilbertson *et al*., 1991; Kubiak and Best, 1991; U.S. EPA, 1993a; Giesy *et al*., 1994; Barron *et al*., 1995; Hoffman *et al*., 1996). Because of this well-established sensitivity, the viability of avian reproduction was selected as an endpoint in this assessment. To determine avian reproductive TRVs, the scientific literature was reviewed to identify doses of PCHs to adult birds known from previous studies to have resulted in adverse effects on fertility and embryo survival. Ideally, such studies would involve the long-term dosing of adult female birds with PCH-contaminated food (or direct dosing) and the subsequent quantification of reproductive performance. The resulting data would be interpreted as:

- Lowest observed adverse effects levels (LOAELs), i.e., the lowest dose rate associated with impaired reproductive performance

- No observed adverse effects levels (NOAELs), i.e., the highest dose rate that did not result in impaired reproductive performance.

Only one avian study of 2,3,7,8-TCDD conformed to the above requirements. Nosek *et al*. (1993) subjected female pheasants to intraperitoneal injections of 2,3,7,8-TCDD over a 10 week period that included reproduction. Three dose levels were included and from these a LOAEL and NOAEL can be established (140,000 and 14,000 pg/kg bw/d, respectively). These body-weight normalized TRVs were converted to NOAEL and LOAEL daily doses for each of the four avian receptor species (using the body mass data in Table 4-1). The results are reported in Table 4-12.

The risks posed to avian reproduction by the ingestion of TCDD-EQ are calculated in this ERA by comparing their TCDD-EQ intakes (Section 4.1.7) to the NOAEL daily doses in Table 4-12.

4.2.2 Insectivorous and Carnivorous Mammals

Sample *et al.* (1996) reviewed the laboratory studies in which mammals were dosed with PCDD/PCDFs. Only one study subjected a mammal (the laboratory rat) to contaminant over a long time period and quantified the effects on reproduction: Murray *et al*. (1979)

subjected three generations of rats to three dose levels of 2,3,7,8-TCDD. Reproductive LOAELs and NOAELs were 0.00001 and 0.000001 mg/kg bw/d, respectively. From these results, Sample *et al*. (1996) derived red fox and short-tailed shrew LOAELs (normalized to body weight) of 0.0000053 and 0.000022 mg/kg bw/d, respectively, and NOAELs of 0.0000005 and 0.0000022 mg/kg bw/d, respectively. Poiger *et al*. (1989, reviewed in Sample *et al*., 1996) dosed laboratory rats with 1,2,3,7,8-PeCDF, 2,3,4,7,8-PeCDF, 1,2,3,4,8-PeCDF, and 1,2,3,6,7,8-HxCDF over a 13-week time period. However, these studies did not focus on effects on reproduction and are, therefore, not used to select TRVs in this ERA.

These body-weight normalized TRVs (expressed as pg/kg/d) were converted to NOAEL and LOAEL daily doses for each of the two mammalian receptor species (using the body mass data in Table 4-1). The results are reported in Table 4-12.

The risks posed to mammalian reproduction by the ingestion of TCDD-EQ are calculated in this ERA by comparing their TCDD-EQ intake rates (Section 4.1.7) to the 2,3,7,8-TCDD NOAEL daily doses in Table 4-12.

Table 4-12. NOAELs and LOAELs identified as toxicity reference values in this ecological risk assessment

Analyte	Species	NOAEL (pg/kg bw/d)	LOAEL (pg/kg bw/d)	NOAEL DAILY DOSE* (pg)	LOAEL DAILY DOSE (pg)
TCDD-EQ	Short-tailed shrew	2,200	22,000	38	380
	Red fox	500	5,000	2,050	20,500
	Red-tailed hawk	14,000	140,000	17,136	171,360
	American kestrel	14,000	140,000	1,820	18,200
	American robin	14,000	140,000	1,134	11,340
	American woodcock	14,000	140,000	3,080	30,800

* used as TRVs in this ERA

5. RISK CHARACTERIZATION

Table 5-1 shows the HIs calculated using the estimated TCDD-EQ daily intake rates of the six receptor species (Section 4.1.7) and the NOAEL daily dose TRVs in Table 4-12. The estimated daily intake rates from Section 4.1.7 are based on the mean concentrations of PCDD/PCDFs in floodplain soils downriver of Midland (Table 4-5). Table 5-1 also shows the corresponding HI values calculated based on the soil median, maximum and upper 95% confidence limits on the mean (95% UCL) TCDD-EQ concentrations (Table 4-11). The calculations were simple proportionalities: for example, if a mean TCDD-EQ soil concentration of 945 pg/g (Table 4-11) results in red fox HI of 845 (Table 5-1) a maximum soil TCDD-EQ concentration of 7,421 pg/g (Table 4-11) will result in an HI of 7,421/945 x 845 = 6,636.

The HIs shown in Table 5-1 vary between 845 and 28 based on the mean soil concentrations, 399 and 12 based on the median soil concentrations, 6,636 and 220 based on the maximum soil concentrations, and 1,191 and 39 based on the 95% UCL soil concentrations.

Table 5-1. Hazard Indices (HI) calculated for six receptor species using soil mean, median, maximum, and 95% UCL PCDD/PCDF concentrations as bases for food chain model.

Species	TRV (pg TCDD-EQ/d)	Exposure (pg TCDD-EQ/d)	HI (mean soil conc.)	HI (median soil conc.)	HI (max. soil conc.)	HI (95% UCL soil conc.)
Red fox	2,050	1,732,613	845	399	6,636	1,191
Short-tailed shrew	38	1,049	28	13	220	39
Red-tailed hawk	17,136	1,586,547	93	39	804	135
American kestrel	1,820	318,013	174	73	1,505	252
American woodcock	3,080	90,586	29	12	251	42
American robin	1,134	46,879	41	17	355	59

The differences among the magnitudes of the HIs in Table 5-1 are partly a function of trophic level: the top predators red fox, red-tailed hawk and American kestrel have the greatest HIs, while organisms that feed lower in the food web have smaller HIs.

The soil PCDD/PCDF concentrations in the assessment area vary spatially and the actual exposure (and, hence, risk) to any receptor is a function of this spatial variability, together with the extent of the area over which it might forage for food. A red fox with a hunting territory of 500 ha is likely to integrate in its exposure the range of spatial variability in contaminant concentrations over that area. Therefore, using the maximum or upper soil 95% UCL concentrations as a basis for calculating exposure and risk to this species may be over-protective, and the mean or median concentrations may be more reasonably protective. The same is true for the red-tailed hawk and the American kestrel. For organisms with smaller home ranges, however, some individuals may be exposed to the higher concentrations of contaminants and suffer higher risk. Thus, it is more reasonable to base the risk estimates to the American robin, the American Woodcock, and the short-tailed shrew on the higher soil concentration estimates. Based on these considerations, Table 5-2 shows what are considered to be the most relevant ranges of HI values for the receptors.

Table 5-2. Most relevant HI ranges for the receptor species.

Species	HI estimate
Red fox	399 – 845
Short-tailed shrew	28 – 220
Red-tailed hawk	39 – 93
American kestrel	73 – 174
American woodcock	29 - 251
American robin	41 - 355

These results show that:

- The HI values for all the receptor species exceed unity, thus risk to any of them due to exposure to PCDDs/PCDFS in the assessment area cannot reasonably be discounted

- The greatest HIs generally apply to the top predators, and the lower HIs apply to organisms that feed lower in the food chain (the insectivores). This conforms to what would be expected given the propensity of PCDDs and PCDFs to biomagnify within food chains

In summary, given the assessment area soil concentrations of PCDDs/PCDFs and the modeling performed in this ERA, risks to at least six species of terrestrial organisms cannot be discounted. Indeed, given the very high HI values calculated, it may be more likely than not that further ERA analyses will confirm the existence of this risk.

6. UNCERTAINTIES

Uncertainty is an intrinsic part of all ecological risk assessments, and indeed of all studies of the effects of stressors on organisms living under uncontrolled circumstances. Even if highly detailed field studies are performed to provide site-specific data, uncertainty still cannot be avoided. Indeed, while reducing some of the original sources of uncertainty, such studies may introduce other sources of uncertainty.

Uncertainty in ERA may arise from a large number of sources but most often because it is usually not possible to accurately predict exposure to all of the potential receptors, or that the stressor response information is not complete and assumptions must be made, or that no stressor-response information exists for the receptor and a surrogate species must be used. Regardless of its source or type, the ERA must, to the extent possible, explicitly recognize and accommodate this uncertainty. If, given the constraints of data availability, it is not possible to entirely eliminate sources of uncertainty, the remaining sources must be brought to the attention of the risk manager.

In this ERA, uncertainty potentially arises from five main sources. These are identified below and their likely impacts on the certainty with which the risk results can be viewed are discussed.

6.1 TRVs

The TRVs shown in Tables 4-12 and 5-1 are based on a relatively small number of studies and study species (one each for mammals and birds). Thus, there is some uncertainty regarding the actual sensitivities of the six receptor species. It is possible that some of them could be less sensitive than the two experimental species (ring-necked pheasant and laboratory rat) and the HIs in Table 5-2 will overestimate the degree of risk. However, it is equally possible that some of them could be more sensitive and the data in

Table 5-2 might underestimate the degree of risk. Even if it were (implausibly) assumed that all of the TRVs used in this ERA overestimate risk, they would have to be reduced by up to 3 orders of magnitude before all of the HIs dropped below 1. Thus, while it is acknowledged that there is uncertainty associated with the TRVs used in this ERA, it would be unrealistic to assign the high HI values to this alone.

6.2 Uptake Factors

Relatively few data have been published in the scientific literature describing food chain uptake of PCDD/PCDF congeners. The few data that are available are shown in Table 4-3. Given this relatively small data set, assumptions have been made in this ERA regarding uptake of specific congeners by wildlife species. Where this has been done, it has been conservatively assumed that the uptake factors would be closer to the lower ends of the likely ranges. This may result in the assumed uptake factors not being protective enough of exposed wildlife species. This conclusion is supported by data that have recently been gathered by Custer *et al*. (in press), who found that the uptake factors between emerged aquatic insects in a riparian area in Rhode Island and nestling tree swallows (*Tachycineta bicolor*) were: TCDD 5-8, TCDF 6-9, 2,3,4,7,8-PCDF 1.5-8.7. While these data do not strictly refer to a terrestrial food chain, they do suggest that that the UFs used in this ERA might be underprotective and, therefore, the risks underestimated.

6.3 Food and Soil Ingestion Rates

The food and soil ingestion rates that were used to estimate receptor exposure in this ERA were derived from the scientific literature, rather than site-specific studies. Nevertheless, the rates that were used represent our best scientific judgment, obtained from reputable peer-reviewed sources.

6.4 Congener Contributions to Risk

The high HI values calculated in this screening-level ERA are largely a function of two congeners: 2,3,7,8-TCDF and 2,3,4,7,8-PCDF, with relatively minor contributions from the other congeners known to be present in the assessment area. Thus, the conclusions of the ERA are sensitive to uncertainty regarding their environmental behavior, particularly their uptake and biomagnification. The UFs selected in this ERA were based on empirical data from other studies and a reasonably protective approach. Therefore, the HI values obtained are robust, given our current scientific knowledge. However, site-specific data would help reduce any uncertainty further.

6.5 Carrion in the Red Fox diet

A substantial proportion of the diet of the red fox is assumed (based on literature information) to be carrion. For the purposes of this ERA it has been assumed that this carrion will be contaminated with PCDD/PCDFs to a concentration midway between that of small mammals and birds. However, it is conceivable that at least some of this carrion could be white-tailed deer, in which case the contamination level might be lower. Even if we assume, however, that the carrion that is eaten by the foxes is entirely uncontaminated, the HI (based on mean soil contaminant concentrations) is reduced to only 547. Therefore, assuming such a low (non-existent) level of contamination in the carrion component of the diet does not result in the estimated risks being brought to an acceptable level.

6.6 Statistical Measures

Screening-level ERAs do not seek to answer questions about the magnitude or risk or its spatial distribution, but address the more basic question: can unacceptable risk be reasonably discounted? For this reason, and because they are typically based on limited site-specific data, which can contribute to the possibility of making false negative decisions, they typically incorporate conservative assumptions regarding exposure to wildlife and the sensitivities of exposed species. Thus, maximum or 95% UCL concentrations in media are typically used to estimate exposure, rather than mean or median concentrations. In this screening-level ERA a reasonable degree of protectiveness has been sought by presenting HIs based on a number of statistical measures (means, medians, maxima and 95% UCLs). No matter which measure is selected, the HIs persistently exceed 1 (Table 5-1) and are often more than 2 orders of magnitude higher than that. Irrespective of the sources of uncertainty identified above, these results unambiguously show that the possibility of unacceptable risk to receptors in the Tittabawassee River floodplain downriver of Midland cannot be reasonably discounted.

7. RISKS ACROSS HABITATS

In this ERA and in the ERA previously carried out for the aquatic environment of the Tittabawassee River (GES, 2003), it has been assumed that receptors in either analysis are exposed only to contaminants within either the aquatic or terrestrial food chains. For many of the organisms this is true: avian piscivores (for example) are unlikely to be exposed to contaminants in small mammals in the floodplain. However, some organisms may be exposed through feeding on prey from both ecosystems. Mink (*Mustella vison*), for example, may switch between aquatic and terrestrial prey depending on local conditions. The risks posed to this organism would then accrue from both habitats.

In GES (2003) the risks to mink feeding on fish prey from the Tittabawassee River were evaluated. As part of this analysis it was assumed that the mink were able to switch their diets to include terrestrial prey, which were assumed to be uncontaminated with PCHs. The result of this analysis was that the mink in the assessment area could escape risk by limiting their fish intake to about 2% of their diet and relying on terrestrial prey from the floodplain for the remainder. This terrestrial analysis has shown, however, that ingesting prey from within the floodplain would also expose organisms that are able to switch between habitats to risk. Only by going outside the floodplain would such organisms, be able to avoid unacceptable risks.

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